Sustainable Animal Agriculture

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Sustainable Animal Agriculture

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Preface

The use of the word ‘sustainability’ has increased tremendously in the last decade. Since the 1980s, there has been an explosion of publications in ‘sustainability science’. Sustainability has been used to describe various systems including aspects of agriculture, business, energy utilization, etc. Although it is a feature that many users rate highly in any given system, its exact definition remains elusive. Thus, this book opens with a discussion of how one can define sustainability and how it is achieved or even if it can be achieved (Chapter 1). There have been numerous definitions offered including by the US Congress and various governmental and non-governmental agencies. At least in agriculture, there is a consensus that the definition offered by the United Nations (Brundtland) Report for sustainable development as ‘development that meets the needs of the present without compromising the ability of future generations to meet their own needs’ (United Nations, 1987) is a workable description. However, it is challenging to put it in practice. For this book, we take the triple-bottom-line approach in which sustainability is found at the confluence or ‘sweet spot’ of environmental, economic and social considerations.

In animal agriculture, there has been an emphasis on the ‘environmental’ aspect of sustainability, which is also reflected in the composition of this book (Fig. 1). For example, Chapters 2, 3, 5, 7, 8, 9, 10, 11, 14, 15, 16 and 17 deal with various aspects of the environment such as water, air and soil issues. This does not mean that the environmental aspect is more important but reflects the amount of resources and research devoted to it compared with other aspects of sustainable animal agriculture. Chapters 2, 3 and 11 for example deal with improving efficiency; however, we should not confuse sustainability with merely improvements in efficiency. The world’s population is growing rapidly, particularly in developing countries. Therefore, more and more food will be required. Growing more food will always have an environmental cost. In the short term at least, total emissions will be growing. What we need to strive for is not to stop growing food but improving efficiency so that we can slow down the rate of emissions and eventually reverse the total amount of emissions through either transformative or incremental system-wide progress. Although higher efficiency would ultimately reduce the carbon footprint per unit product and may be viewed as environmentally sustainable, it does not mean it leads to a more sustainable system overall when constraints and other considerations are taken into account. This is one of the reasons why comparisons of different production systems are challenging. For example, grazing may have higher carbon footprint than concentrated feeding operations but on marginal lands or places where the land can only be used for pasture, it is the most sustainable way of using that particular land. So when I hear the passionate views about

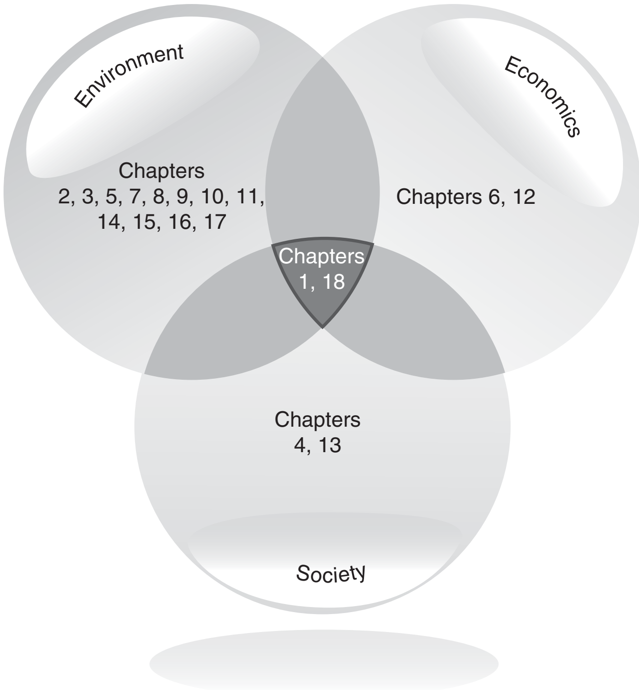

Fig. 1. The triple-bottom-line approach to sustainability and how various chapters in this book address issues surrounding sustainable animal agriculture. (Drawing courtesy of Suban Foiklang.)

whether farming should be entirely organic and free of any genetically modified organisms, or entirely industrial-scale with the efficiencies of synthetic tools, my response is that it will be all of these things.

According to the Intergovernmental Panel for Climate Change (IPCC), warming of the climate system is unequivocal and it is very likely (>90% probability) it is due to human activity (IPCC, 2007). Rockström *et al.* (2009) further showed that climate change, biodiversity and nutrient loading is already operating in a dangerous zone and animal agriculture has an impact on all of these matrices, thus creating a need to come up with a more sustainable solution. Sustainability can be viewed as a solution to a linear programming problem (Chapter 6) where there will be compromises in various aspects of the system and it is about cooperation and dialogue that will help to achieve the best or highest use of available resource for that particular land or region subject to constraints.

In animal agriculture, social issues have not been given enough consideration. An attempt is made to explore what society might expect from a more sustainable system (Chapter 13). In addition, how changes in animal agriculture affect farmers, ranchers and farm workers is also discussed. If animal welfare can be seen as belonging to the 'social' category, Chapter 4 gives details of how scientific study of animal welfare issues can be used to inform the public and lead to a more sensible regulation that takes into consideration various priorities of stakeholders.

More sustainable systems are likely to develop through implementation of a number of small mitigation options rather than one big alternative or 'silver bullet'. However, we may need a transformational change in the way animal agriculture (in conjunction with other agricultural activities) is conducted. There is a need to develop a 'road map' with inputs from various disciplines of natural science and humanities to allocate resources (research and outreach) in order to develop a more

sustainable system. Chapter 18 focuses on what is next for sustainable animal agriculture and the concept of 'back to the future', i.e. mixed farming as a tool to a more sustainable system, is explored.

Finally, sustainability should be viewed as a journey, not a destination. It has multiple goals, and therefore trade-offs and sometimes co-benefits will occur. Maximizing multiple goals including environmental, social and economic aspects of food production will lead us to a more sustainable system.

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1 Sustainability: a Wicked Problem

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Introduction

Sustainability is a product attribute that many consumers appear to want today, even if it is a distinctly different characteristic from more traditional product attributes such as freshness and taste. Many large global food corporations as well as many smaller ones have appointed sustainability officers to their senior management in recent years. Among the 50 largest global food and beverages companies, 23 have created or joined various types of multi-stakeholder engagements in pursuit of enhanced sustainability. For all of this activity and interest, sustainability remains an elusive term. It is also not clear how it is achieved or even whether it can be achieved. Yet certain stakeholders have passionate positions on the issue and are willing to exercise veto power over traditional market transactions in its name. For the dairy industry in particular, one need only mention such examples as the demise of rBST, the rise of animal welfare initiatives and opposition to concentrated animal feeding operations (CAFOs). Producers in commercial-scale, conventional agriculture often take offence at so-called 'sustainable agriculture' that has come to include organic and local but seems to exclude them as 'non-sustainable', at least by implication. Sustainability is further used by many as a code word for environmental concerns being the chief criterion in natural resource decision making, in supposed contrast to corporate resource users' concern for profit to the exclusion of any other criteria.

The purpose of this chapter is to frame sustainability as one type of 'wicked problem' that cannot be solved, only managed. The framing is useful for several reasons. First, it helps explain why there are so many varying definitions of sustainability and what useful working definition can be adopted. Second, the passionate discord among relevant stakeholders is to be expected with such problems and cannot be dismissed but rather must be managed. Third, new knowledge is especially critical to managing wicked problems and thus scientists and other knowledge workers are critical to altering the trajectory of food and agriculture systems to be more sustainable. But the role of scientists is dramatically different in the world of wicked problems than in the world of tame ones. Finally, active engagement of all stakeholders in co-creating system innovation is one of the few (if only) ways forward toward sustainability. This engagement demands transdisciplinary scholarship on the part of university scientists if they are to be effective contributors to managing wicked problems.

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The Problem with Wicked Problems

A wicked problem (Rittel and Webber, 1973; Conklin, 2006) is a term from the 1970s social planning literature. *Such a problem has the essential characteristic that it is not solvable; it can only be managed.* The combination of Rittel and Webber (1973) with Conklin (2006) provides a lengthy list of relevant criteria that characterize wicked problems. Four of these criteria are adopted here as an efficient set to define the concept:

- No definitive formulation of the problem exists.
- Its solution is not true or false, but rather better or worse.
- Stakeholders have radically different frames of reference concerning the problem.
- The underlying cause and effect relationships related to the problem are complex, systemic and either unknown or highly uncertain.

Sustainability fits these criteria quite well as the analysis will show. Fuel versus food, climate change, poverty alleviation and even business strategy (Camillus, 2008) are other examples.

No definitive formulation of the problem exists

The concept of sustainability has at least one simplistic and intuitive definition: using a resource today is sustainable if it does not constrain the use of the resource tomorrow. This definition paraphrases the definition in the well-known United Nations (Brundtland) Report 'Our Common Future' – sustainable development 'meets the needs of the present without compromising the ability of future generations to meet their own needs' (United Nations, 1987). Consuming non-renewable resources is not sustainable precisely because their use today diminishes their supply for use tomorrow without any reasonable prospect for replenishment. The definition and the example seem very clear. The trouble arises through the realization that by this definition there is probably no use of resources that is sustainable. Human use of resources is always less than 100% efficient, and the finite bounds of the earth suggest that eventually every resource will be exhausted. Any economist takes as given the foundational notion that resources are fundamentally scarce and the study of their efficient use is thus essential. Sustainability as the simple conversion of resources is thus unattainable in a realistic way or undesirable if produced solely by subsistence living. Moving beyond this simple definition leads to an endless debate about what sustainable is or is not. Sustainability has no definitive formulation as to what it means and how it is achieved.

Its solution is not true or false, but rather better or worse

In the place of a definitive formulation, various attempts at a working or implementable definition arise. Today these more workable definitions include:

1. The triple bottom line. Something is sustainable if it can simultaneously achieve economic feasibility, social responsibility or justice, and environmental quality.
2. The three Ps – Profit, People and Planet. This is just another version of the simultaneous criteria for achieving better economic, social and environmental outcomes.

The justification for using such definitions arises from the notion that a balanced set of decision criteria among economic, social and environmental ends changes the trajectory of resource use toward greater sustainability. Sustainability becomes a matter of better or worse rather than true or false – meeting the second criterion of a wicked problem. However, decision makers (public or private) are not given much content by these definitions as to how one achieves the triple bottom line or 3P value enhancement. Progress is a matter of altering trajectories along three paths into the future rather than a tangible single thing to be attained. Attainment is even more muddled by the extent to which the bundle and mix of resources used across time changes – the stone age gives way to the iron age and then to steel, steam gives way to electricity, analogue communication gives way to digital. The constraints

on resource use are continually changing with time and thus the 'solution' to sustainability is forever in flux.

Stakeholders have radically different frames of reference concerning the problem

Now add to the mix multiple perspectives, values and frames of reference from the stakeholders who care about sustainability. Business stakeholders will strongly opt for economic gain and profit first; environmentalists for environment and planet; and social advocates for social and people-oriented outcomes, including justice, fairness and caring. Each stakeholder group has incentive to define sustainable by its own vocabulary consistent with its own end goals. Furthermore, they will each be driven to act on their own agenda – promoting their view of the problem and its 'solution' while working to veto the actions of the other stakeholders as those actions are strongly believed to be in error. Many agricultural production situations come to mind. Manure odour problems are an example – the livestock producer views odour as part of the normal process of agriculture and defines its presence as necessary to production; suburban neighbours in response act to create ordinances that limit (or eliminate) odour, threatening the producer economically.

Part of understanding the consequences of wicked problems is that the range of stakeholders can become quite large. These stakeholders may include non-governmental organizations (NGOs) advocating solutions (e.g. Greenpeace, World Wildlife Federation), governmental bodies having a say in regulation and third-party certifiers acting as arbiters in the process. Knowledge institutions (e.g. universities and think tanks) may also play a role in defining the problem or various options for solution. The key takeaway is that the context in which a wicked problem occurs creates the set of stakeholders, and this set may go well beyond the traditional notion of supply chain customers and suppliers. If a stakeholder group can take action to enable or veto efforts related to the problem, then they are at the decision making table whether other stakeholders want them to be or not.

The underlying cause and effect relationships related to the problem are complex, systemic and either unknown or highly uncertain

This is the fourth and final criteria for a wicked problem. Could anyone meaningfully challenge the following statement: the agriculture-food-bio system that has emerged globally is highly complex; it is a system with multiple interrelated input, output and feedback loops; and, for all that is known from food and agricultural sciences (natural and social), there remain many uncertainties and unknowns about cause-and-effect relationships? This context of system complex ambiguity heightens the impacts of the other three criteria. Stakeholder value judgments cannot always be addressed with clear 'facts' about the system; what constitutes 'better or worse' can even be debated, let alone 'true' or 'false'; and, no definitive definition exists for sustainability to guide the resolution of ambiguity about the system.

Consider this claim that is often made (explicitly or implicitly) about 'sustainable agriculture': 'Small, local, organic production is sustainable while conventional agriculture is not.' Without taking sides, an objective appraiser sees many unresolved system complexities on both sides of this statement. From an environmental perspective, conventional agriculture has substantial sustainability issues based on its intensive use of limited inputs and concentration of potential pollutants. At the same time, conventional agriculture's economic and social (affordable food for the many) impacts have been positive for feeding dramatic increases in global population. But the social impacts are uneven – many hungry people remain in the world while obesity has become a growing health concern. In contrast, small-scale local production can be managed to lower input intensity and to enhance local consumer appeal, but the economic viability of many of these farms remains marginal. Even if this model were superior in some sustainability dimensions, scaling up this model to feed the world would mean finding millions of new farmers to replace the commercial scale farms of today – a substantially new challenge to social and economic sustainability. The complex ambiguity of the system suggests that all stakeholders should carefully weigh their charges and countercharges about what is sustainable.

Sustainability fits all four criteria for a wicked problem quite well. A summary of the analysis is provided in Table 1.1. Further, it becomes rather obvious why wicked problems cannot be solved – they have no closed-form definition, their 'solution' can only be thought of in relative terms, stakeholders will be in conflict over solutions and actions, and the system is not understood well enough to effect entirely purposeful change. In a world that wants simple, implementable solutions, sustainability is unsolvable in any conventional sense. What does one do then with a wicked problem such as sustainability?

Managing Wicked Problems

The literature on wicked problems suggests that while they cannot be solved, they can be managed. The trajectories of the system outcomes – profit, people, planet – can be altered in the short-run to create improved outcomes in the long run. The key is to understand what to manage and how to manage it. (For ease of exposition, the 3P or Profit–Planet–People working definition of sustainability is adopted for the remainder of the chapter.)

The four criteria – no definitive formulation, better or worse trajectories for system outcomes, conflicting stakeholders, and system ambiguity – give insights into managing wicked problems. First, two sets of outcomes need to be managed in the situation:

- *System outcomes.* System components – profit, people and planet – need to be moved in desired directions; their trajectories need to change for the better.
- *Process outcomes.* Relevant stakeholders need to engage in the process, and they need to participate in such a way that they enable system change on the positive side while not exercising their vetoes on the negative.

Consider what happens when one or the other of these two outcomes is not properly managed. In the one instance, potential options to improve the system outcomes will fail to be implemented if the process results in stalemate or dissolution with the offended stakeholders exercising their veto, e.g. taking the debate public in a publicity war or seeking governmental prohibit in law or regulation. In the other instance, solely focusing on process outcomes can devolve into endless process (either unresolved debating or overly polite avoidance of issues that truly divide) with no action taken to improve the system. System and process outcomes must be achieved together if the wicked problem is to be managed.

Any project or process designed to manage a wicked problem would need to begin with

Table 1.1. Sustainability as a wicked problem. Source: Peterson (2011).

Criteria for a wicked problem	Sustainability
No definitive formulation of the problem exists.	Ideal definition lacks specificity and is reduced to slogan or tagline such as triple bottom line (economic, social and environmental) performance.
Its solution is not true or false, but rather better or worse.	One can never know whether sustainability has been achieved. Only progress in its trajectory can be predicted.
Stakeholders have radically different frames of reference concerning the problem, and are often passionate in their position on the problem.	Businesses strongly favour economic outcomes. Environmental groups strongly favour environmental outcomes. Social justice groups strongly favour social outcomes, such as fair wages and equitable access.
System components and cause/effect relationships are uncertain or radically changing.	Many claims are made about what is sustainable (such as local food systems are sustainable while global food systems are not) with unclear knowledge of what system characteristics assure or even promote sustainability.

establishing goals for both types of outcome. On the systems side, the goals should clearly indicate what movements in profit, people and planet are targeted in the system. On the process side, the goals should specify who is at the table and how they will engage together.

Even if this simultaneity of system and process outcomes is a necessary condition for managing wicked problems, it is not a sufficient condition. The roles of new knowledge and innovation management are argued to be the other two necessary conditions. New knowledge and innovation management are critical to resolving (at least to the extent needed for change) stakeholder conflicts and system ambiguity – the other two major concerns arising from the four criteria of a wicked problem.

The Role of New Knowledge

The next step in the analysis begins with an exploration of the meaning of knowledge in this context. The knowledge management literature is especially useful here, particularly the classic work by Takeuchi and Nonaka (2000). Based on this literature, knowledge is about beliefs and commitments, action toward some end, and meaning that is context-specific and relational. In this sense, *knowledge is justified true belief on which an individual or individuals are willing to act.* This concept of knowledge is more Eastern than classic Western, which defines knowledge as truth in some objective, abstract sense. It is relevant to wicked problems because it better explains the relationship that each stakeholder has with existing knowledge that they bring to a situation. Each set of stakeholders view their knowledge as true – justified true belief – and will take action (to enable or to veto). Existing knowledge can be further divided into: (i) *tacit knowledge*, which is justified by being embedded in experience and specific context arising from practice; and (ii) *explicit knowledge*, which is justified by formal documentation and testing (Takeuchi and Nonaka, 2000). Scientific knowledge is an excellent example of explicit knowledge while the knowledge of a long-term, experienced dairy farmer is a classic example of tacit knowledge. Note how the justification of true belief differs between the two types of knowledge.

The existing knowledge that each stakeholder brings to the management of a wicked problem sets up the process for failure. Existing knowledge is deficient to support the management process in two respects. This first deficiency with existing knowledge is the issue knowledge legitimacy. Each set of stakeholders will suspect the knowledge being brought by the other stakeholders because of issues arising from trust, transparency and credibility of sources. On the one hand, each set of stakeholders would likely claim that its explicit knowledge is enough in itself to solve the dilemma – the problem is in fact not wicked. Each stakeholder asks the others to accept their solution, and the problem is solved. Invoking ‘good science’ (the ultimate authoritative explicit knowledge in the western world) is an example of this approach. There is much anecdotal evidence that this approach rarely works with wicked problems. Stalemate with duelling scientists is the much more likely outcome. On the other hand, if existing explicit knowledge is not invoked or is repudiated, then one would expect a similar claim that existing tacit knowledge is somehow enough to solve the problem. Various cases or archetypal examples of sustainable or potentially sustainable practices would be invoked by one or another stakeholder. The problem here with sustainability is that common knowledge of such practices is not broadly held and would again be suspected by other stakeholders. There are examples of conservation practices, co-generation-of-energy practices, organic practices, benign environmental practices or fair-trade-for-social-justice practices. But none of these has been shown to be sustainable in regard to the 3Ps or any robust notion of sustainability. Existing tacit knowledge is woefully lacking.

The second deficiency with existing knowledge is the issue of fixed or frozen trade-offs. Existing knowledge, explicit or tacit, has fundamentally led to the value conflicts that separate the stakeholders in the first place. Existing knowledge of the system in question and its use in practice has resulted in the unfavourable trade-offs in profit, planet and people that set the various stakeholders at odds with each other. If the process of managing sustainability simply trades on existing knowledge, then little positive change can be achieved. Based on existing knowledge, one might make minor improvements in

the trade-offs, or one stakeholder group or another might compromise their views to see this or that trade-off as more or less troublesome. It seems hard to imagine that either of these circumstances would bring any significant change in the sustainability trajectory.

If existing knowledge is deficient, then it follows that new knowledge and only new knowledge has the ability to assist in managing wicked problems such as sustainability. However, following the arguments just made, new knowledge only works if two things are true: (i) the new knowledge is developed or discovered in such a way that it is credible and legitimate to all stakeholders (meets each stakeholder's criteria for justified true belief); and (ii) it reshapes the trade-offs among the system outcomes that divided the stakeholders in the first place.

Co-creating New Knowledge and Innovation

As scientists, we are regularly engaged in the process of creating new knowledge, but we need to be careful not to fall into a trap that there is only one source of new knowledge. There are a number of approaches to creating new knowledge, and one suspects that the various stakeholders in a wicked problem will be well aware of this.

Takeuchi and Nonaka (2000) argued that new knowledge is created by various forms of 'conversion' between existing tacit and explicit knowledge. Tacit knowledge becomes new explicit knowledge through the process of *externalization*, i.e. taking what is known from practice and experience and making it formal and accessible to others. Scientific induction is a form of externalization. Tacit knowledge is converted into new tacit knowledge through the process of *socialization*, i.e. sharing experiences and practice with others. Apprenticeship is an excellent example of socialization. Explicit knowledge becomes new tacit knowledge through a process of *internalization*, i.e. taking explicit knowledge (from others) and applying it in one's own practice. Finally, explicit knowledge is converted to new explicit knowledge through *combination*, i.e. combining or synthesizing different bodies of explicit knowledge. Scientific deduction fits here. Existing knowledge can be used to create new knowledge, but it is the new knowledge that is useful in managing wicked problems and the question of legitimacy is not entirely answered if the links to existing knowledge are problematic for some stakeholders.

A second source of new knowledge is developing a new means of 'justifying true belief'. New knowledge that could unfreeze the system trade-offs through breakthrough or new-to-the-world innovation is most likely to come from outside the existing base of justified true belief. Paradigm shifts (Kuhn, 1962) or other radical changes in prevailing mental models will be the source of high value new knowledge. The goal of any new paradigm or significant system innovation would focus on converting existing system trade-offs into complements. Redesign the system to produce profit, planet and people outcomes in complementary ways inside of in competing ways.

Whatever the source of new knowledge, recognize that when first discovered it is neither explicit nor implicit. New knowledge does not have the record of testing and documentation to justify it explicitly while there is no body of practice to justify it implicitly. New knowledge is justified in its early discovery by reference to existing knowledge or paradigms, or by intuition that it makes good sense, it moves us in a more effective direction, or it is useable in practice.

Shift then to the process of creating new knowledge by whatever means (conversion of existing knowledge or new paradigms for system innovation). Knowledge legitimacy and the unfreezing of system trade-offs need to result from the creation process. The process has legitimacy when the new knowledge is derived together with the stakeholders. Further, when the creation of the new knowledge centres on system innovation, then more acceptable impact trade-offs can emerge even to the point of converting existing trade-offs into new complements through deeper systems understanding and redesign. The conditions for managing wicked problems become:

1. Attending to both system outcomes and process outcomes.
2. Co-creating new knowledge with stakeholders to assure knowledge legitimacy.
3. Focusing the new knowledge on system innovation that transforms the system trade-offs into complements so far as possible.

Transdisciplinary Scholarship

Knowledge institutions and their scholars have a role in managing wicked problems like sustainability when they understand how research can be beneficial to the process outcomes as well as the more traditional system outcomes. Research in the wicked problem context cannot simply be undertaken in the matter of normal science (Kuhn, 1962; Batie, 2008) – stakeholders articulate a problem and possible causes with relevant scientists; the scientists go off to their labs or other experimentation places to analyse, create and test solutions; and, finally return with the solution for the stakeholders to implement. The linear model of science has limited application. The problems and causes are too complex for once-and-done definition, and the time away for experimentation gives rise to issues of transparency, stakeholder understanding and commitment, and knowledge legitimacy.

The linear model needs to give way to far more messy process of transdisciplinary research. Multiple disciplines will be required to create the knowledge and innovation for changing the trajectory of the system and process outcomes. All faculties of a knowledge institution will be needed from natural science, social science and humanities. Wicked problems force those of us in the academy to go beyond multidisciplinary approaches (pooling individual disciplinary knowledge) demanding instead transdisciplinary approaches (collective disciplines creating new knowledge together). Transdisciplinary research has power to unite the knowledge actors while drawing upon and transcending individual disciplines.

But transdisciplinary research is not enough. The situation calls for moving to full transdisciplinary scholarship by combining transdisciplinary research with transdisciplinary outreach and education *and* engagement with stakeholders as peers throughout the process. The stakeholders need to be engaged throughout the research enterprise in order to have its results be meaningful and legitimate to the desired process outcomes. The stakeholders cannot merely be there at the beginning of the process (to articulate their needs) and at the end of the process (to receive the results). They must be there throughout the process to assure that the research stays on track and will have stakeholder credibility when the results are known. The researchers will need to manage the rigour of the research, but the research will be done in a fishbowl unlike our traditional research expectations of objective separation.

Transdisciplinary scholarship that encompasses all of the above is essential when working in the arena of wicked problems. This realization is entirely consistent with the historic and contemporary literature that surrounds wicked problems and sustainability. (See Fear *et al*. (2006), Batie (2008), Bitsch (2009), Thompson (2010) and Peterson (2011), for contemporary analyses related to agricultural and natural resource systems.)

Two Dairy Examples

Two examples are offered of multi-stakeholder innovation processes that have in their own way created changes in the trajectory of dairy sustainability. The examples demonstrate that real world actors are working with the concepts that are laid out in this chapter. Many other examples likely exist, but they show in a ‘boots on the ground’ sense how to manage (not solve) sustainability. The author has extensive personal knowledge of both examples in addition to specific source material including the SAI website (SAI, 2012) and the book *Sustainable Agricultural Entrepreneurship* (van Altvorst *et al*., 2011).

The first example is the Sustainable Agriculture Initiative (SAI) Platform developed in 2002 by Nestlé, Unilever and Danone. Since its founding, SAI has grown to some 40 members that represent stakeholders throughout the global food supply chain. The unique characteristics of SAI include:

- Being the only global food industry initiative for sustainable agriculture;
- Seeking involvement from all food chain stakeholders;
- Gathering and developing knowledge on sustainable agriculture which is then openly shared with all interested parties;
- Pursuing an inclusive approach to knowledge development from any valuable initiatives and projects from whatever legitimate source, including integrated and organic farming; and
- Implementing concepts and best practices through a continuous improvement process.

The use of new knowledge, innovation and multiple stakeholders is at the heart of SAI. SAI carries out its mission through commodity-specific working groups – coffee, dairy, fruit, arable and vegetable crops, and water and agriculture. In particular, the dairy working group includes an array of formidable global dairy companies: Arla Foods, Cayuga Marketing, Danone, Delaval, Fonterra, Friesland Campina, General Mills, the Innovation Center for US Dairy, Kraft Foods, McDonald's, Nestlé, Novus, PepsiCo and Unilever. The group has established many relationships and alliances around the world. It has developed, published and implemented educational programmes related to their comprehensive *Principles and Practices for Sustainable Dairy Production*. The group has worked on resolving issues of how to measure sustainability at the global level while implementing local programmes of sustainable dairy practice in as diverse locations as Mexico, Chile, Pakistan and India.

The second example comes from the Netherlands and was a project funded by the Dutch innovation programme called TransForum. The project was named The Northern Friesian Woods. The agriculture in this region of the Netherlands is dominated by dairy production. Some sample statistics give a snapshot of the region: 850 farmers with 40,000 ha of land, 148,000 residents, 1700 km of wood banks managed by farmers and 300 million l of milk production a year (2.7% of the Dutch total) generating income of €91 million. The Association of Northern Friesian Woods had as its mission: 'The development of agriculture and the regional economy, in conjunction with strengthening the cultural–historical landscape and its ecological features.' They created a multi-stakeholder coalition that included government, several universities and NGOs to implement a comprehensive programme to carry out the mission.

New methods of production and new products grew from the work. The programme encompassed regional branding for tourism and value-added dairy products, harmonization of agriculture with the need to protect the National Landscape designation for the region, developing energy sources in the form of wood pellets from the previously underutilized wood banks, and linking the attributes of sustainability and closed-loop farming to their marketing of value-added products. All three Ps of sustainability show simultaneous enhancements with the unified work of the region's participants.

Much more could be said about both examples. The intent here is to show that the concept of managing sustainability productively can be achieved.

Takeaway Concepts

Sustainability, be it in dairy or any other dimension of food and agriculture, can best be understood through the lens of wicked problems – ambiguously complex, value-laden and not solvable in any conventional sense. We seem to be in a time in which many problems are becoming increasingly wicked, as value-diverse stakeholders are increasingly willing to exercise their veto power over what industry does. Wicked problems and figuring out how to manage them will be more relevant to what knowledge institutions, governments, businesses and societal organizations must do.

New knowledge is critical to managing wicked problems. Only new knowledge can overcome the twin barriers that existing knowledge faces: (i) lack of legitimacy across relevant stakeholder groups; and (ii) 'frozen' trade-offs among key system components. New knowledge creation requires multiple stakeholders co-creating system innovation through aligned incentives and active experimentation in practice and in scholarship. New knowledge co-created has stakeholder legitimacy, and when it is focused on system innovation, it has the potential to turn trade-offs among profit, planet and people into complements

To play a role, knowledge institutions and their faculty must practise *transdisciplinary scholarship* combining transdisciplinary research with transdisciplinary outreach and education that engages stakeholders as peers throughout the innovation and wicked problem management process.

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2 Production Efficiency of Ruminants: Feed, Nitrogen and Methane

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Introduction

The world population is expected to grow from 6.9 billion in 2010 to 9.3 billion in 2050 (United Nations Population Division, 2010). Less developed countries will experience the most population growth, with only minor population increases in developed countries. By 2050, this expanded world population is expected to consume two-thirds more animal protein than it does today. Total consumption of meat is projected to increase by 73% and that of dairy products by 58% (FAO, 2011). The prospects to increase significantly the amount of arable agricultural land are small, and food production must intensify to ensure an abundant food supply.

Increased animal production has the potential for added consequences (usually perceived to be negative) for the environment, unless steps are taken to ensure that the natural resource base (land, vegetation, water, air and biodiversity) can be sustained while still increasing food production. Thus a major challenge is not merely increasing productivity, but achieving this increased productivity without adverse consequences (Godfray *et al.*, 2010). Conversion of feed into milk or meat is associated with various energy and nutrient losses, including excretion of nitrogen (N) and phosphorus (P) in faeces and urine, and emission of environmental pollutants including the greenhouse gases methane (CH_4), nitrous oxide (N_2O) and carbon dioxide (CO_2). Moreover, resources such as arable land, water, minerals and fuel are limited. The livestock sector is by far the largest anthropogenic user of land. Overgrazing and erosion caused by livestock action is a major contributor to land degradation. On the other hand, livestock production makes important contributions to agricultural economies throughout the world. Livestock products provide a major share of food for human consumption, and livestock production is socially and politically significant, contributing to livelihood and employment of many people (Steinfeld *et al.*, 2006). Mearns (2005) argues that livestock production can play an instrumental role in supporting sustainable rangeland management, preserving wildlife and other forms of biodiversity, and enhancing soil fertility and nutrient cycling. In particular, since the area of grazing land is more than double that of cropping land (FAO, 2011),

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the ability of ruminants to turn human-inedible products into human-edible products is of large significance in terms of global food security.

Increases in livestock production will largely have to come from further advancement in the efficiency of livestock systems in converting human inedible resources into human edible food whilst at the same time improving animal performance. This review will focus on the production efficiency of cattle, primarily that of dairy cattle, at the animal level. Improvements in production efficiency are discussed with respect to the amount of feed required for production and associated losses of N and CH_4 from cattle.

Efficiency of Converting Feed to Product

In general, efficiency of cattle production is concerned with minimizing the amount of inputs (including feed) and losses of nutrients or undesired outputs (e.g. CH_4 and NH_3) to produce a given quantity of meat or milk. Improved efficiency may come from maximizing production (milk production by dairy cattle or growth rate of beef cattle) or from minimizing waste output. The efficiency of feed use for milk or meat production has obvious economic impact, and is the major determinant of feed conversion efficiency (FCE, the amount of product produced per kg of feed dry matter intake; Reynolds *et al.*, 2011). Producers have continued to improve the efficiency with which cattle convert feed into meat and milk through development and application of new technologies and methodologies. In the conversion of feed energy to milk energy, several steps may be distinguished (see Fig. 2.1 for major steps in milk production). Each of these steps in the conversion of feed to milk occurs with varying efficiency, and represents a potential opportunity for improvement.

Not all feed eaten by cattle is useful for production because some of it is not digested but rather lost in faeces. Feed digestibility varies widely, in general with high digestibility values for concentrate feed ingredients rich in non-structural carbohydrates, and low values for roughages high in structural carbohydrates. Numerous factors affect feed digestibility, including level of intake, retention time in the various gastro-intestinal compartments, and feed processing methods (Mertens, 2005; Van der Poel *et al.*, 2005). Although concentrate, and in particular cereal grains, contains high levels of non-structural carbohydrates and have a high digestibility compared with most forages, their rapid fermentation in the rumen may lower pH and decrease digestibility of fibre (Zebeli *et al.*, 2008; Dijkstra *et al.*, 2012).

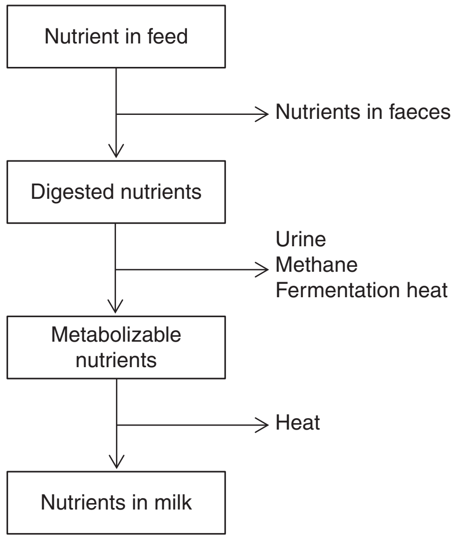

Fig. 2.1. Steps and losses in the conversion of feed into milk in dairy cattle.

Next, some of the digested feed is lost as gaseous energy, primarily CH_4 produced during ruminal and to a lesser extent hindgut fermentation, and as urinary energy, primarily urea produced during the catabolism of organic molecules containing N. Production of CH_4 and urinary N losses are discussed in subsequent sections.

Of the remaining digested feed available for metabolism, some is used for body maintenance purposes, some is lost as heat increment associated with the work of digesting and metabolizing nutrients, and some is converted into milk, accreted body tissues and conceptus. In particular, liver and gut may account for a large and variable part of nutrient metabolism (Lapierre *et al.*, 2006; Reynolds *et al.*, 2011) and contribute to maintenance requirement and nutrient delivery to the various body tissues or organs. At the tissue or organ level, nutrient utilization depends upon the nature and quantity of the absorbed nutrients as well as the physiological

and hormonal status of the animal. Maintenance of body function has a high priority. This requires in particular the supply of ATP via oxidation and the supply of amino acids for obligatory body protein turnover. The maintenance requirements of animals are usually considered fixed. High production levels will increase FCE simply because the amount of metabolized feed needed for maintenance processes is diluted out (VandeHaar and St-Pierre, 2006), as discussed in the next section. The efficiency in which absorbed nutrients are converted into milk or meat also depends on the type of nutrient absorbed. For example, the efficiency of use of acetic acid appears to be lower than that of most other absorbed nutrients, although acetate may be used with higher efficiency if the supply of glucogenic nutrients is high (Tyrrell *et al.*, 1979). To reflect such variation in efficiency, in various feed evaluation systems the efficiency with which metabolizable energy is converted into net energy depends on feed metabolizability (the ratio of metabolizable energy to gross energy), although the effect of metabolizability on this efficiency may well be considerably smaller than assumed in present feed evaluation systems (Strathe *et al.*, 2011).

Production Level and Efficiency

Production level is among the most significant elements in cattle production that determine FCE. An animal that eats more feed to support greater production partitions a smaller portion of feed towards maintenance needs and a greater portion is transferred to the product (milk, meat, etc.) (Fig. 2.2). This improvement in productive efficiency is the classic dilution of maintenance effect, by which the total resource cost per unit of product is reduced (Bauman *et al.*, 1985; VandeHaar and St-Pierre, 2006). The nutrient requirement of cattle comprises a basal, specific quantity needed to maintain the

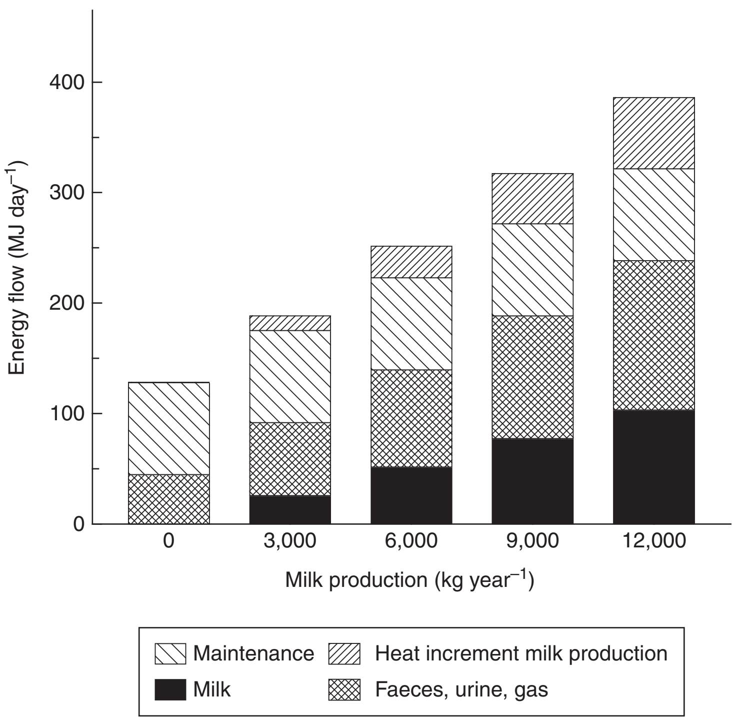

Fig. 2.2. Milk production level (kg fat and protein corrected milk per cow per year) and partitioning of feed towards losses in faeces, urine and gases, maintenance needs, heat increment for milk production, and milk production in dairy cattle. Values based on Feed into Milk system (Thomas, 2004).

vital functions and minimum activities (maintenance requirement) of the animal plus extra nutrients to support the cost of growth or lactation. Thus, upon an increase in production level, less feed resources per unit product are required. This is illustrated in Figs 2.3 and 2.4. In Fig. 2.3, requirements of energy per unit of milk as a function of production level in dairy cattle are shown using a net energy (NE) system (the Dutch NE for lactation system; Van Es, 1978) and a metabolizable energy system (the UK Feed into Milk system; Thomas, 2004). Similarly, the requirements for metabolizable protein per unit of milk in the Dutch (Van Duinkerken *et al.*, 2011) and UK (Thomas, 2004) systems are shown in Fig. 2.4.

Because the maintenance energy requirement is assumed not to change as a function of production, whereas the daily energy requirement increases as milk yield increases, the proportion of total energy used for maintenance is reduced. For example, upon an increase in annual fat and protein corrected milk (FPCM) production from 6000 to 10,000 kg per cow, the energy requirements per kg milk (MJ kg^{-1} FPCM) are reduced by 16% and 19% in the Dutch and UK systems, respectively (Fig. 2.3). Similarly, as described more extensively in Chapter 11 of this book, Capper *et al.* (2009) and Capper (2011) showed that modern, high-production level dairy and beef cattle practices require considerably fewer feed resources than low-production level systems several decades ago. Average annual milk production in the USA increased from 2074 kg per cow in 1944 to 9193 kg per cow in 2007 and feed input per kg milk was reduced by 77% at the 2007 level compared with the 1944 level. Average daily growth rate of beef cattle in the USA increased from 0.75 (1977) to 1.08 (2007) kg per head, and feed input per kg gain was reduced by 19% at the 2007 level compared with the 1977 level. Although maintenance requirements are generally assumed to be proportional to metabolic body weight, it should be noted here that some organs or tissues, in particular visceral organs, have high metabolic rates and are responsive to altered nutrient intake (Baldwin *et al.*, 1985). Possible differences in weights of those organs or tissues or in their response to altered nutritional states and production levels may actually lead to some variation in maintenance energy requirement. The partial efficiency of converting ME to NE in dairy cattle appears

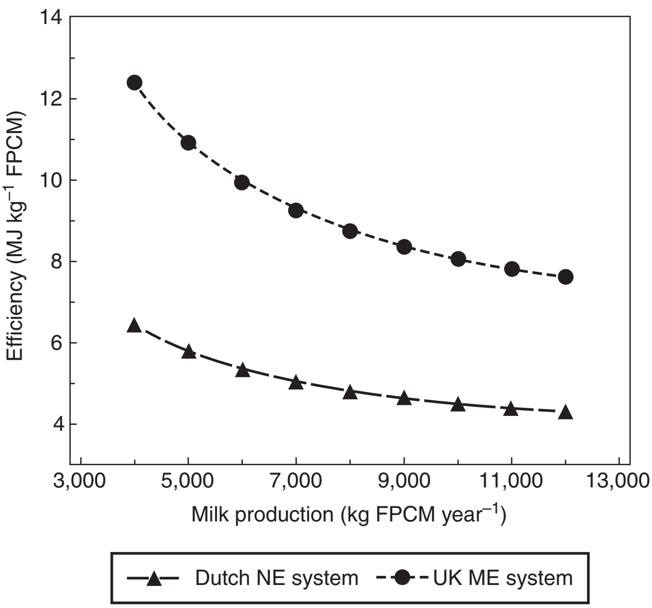

Fig. 2.3. Efficiency of conversion of net energy (NE) or metabolizable energy (ME) (MJ) into milk (kg fat and protein corrected milk (FPCM) per cow per year) according to the Dutch NE system (van Es, 1978) and the UK Feed into Milk system (Thomas, 2004).

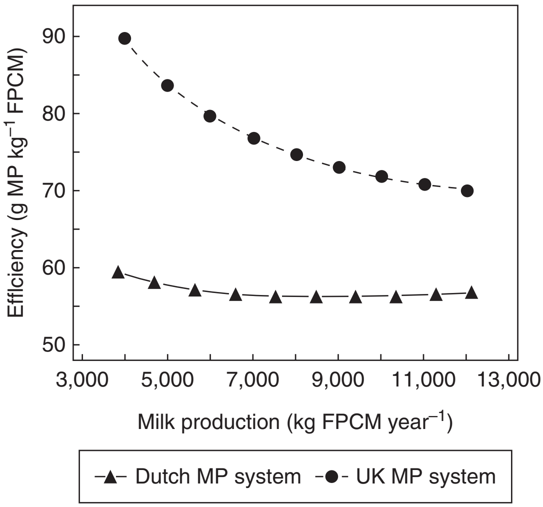

Fig. 2.4. Efficiency of conversion of metabolizable protein (MP; g) into milk (kg fat and protein corrected milk (FPCM) per cow per year) according to the Dutch protein evaluation system (van Duinkerken *et al.*, 2011) and the UK Feed into Milk system (Thomas, 2004).

largely constant across a wide range of ME intakes, and there is no statistical reason to suggest that feeding level affects this efficiency (Kebreab *et al.*, 2003).

Although the amount of feed required per unit of milk or meat is reduced when milk production levels increase, the associated higher feed intake levels generally coincide with a reduction in digestibility. The level of reduction in digestibility is usually more pronounced for structural carbohydrates compared with non-structural carbohydrates. For example, Robinson *et al.* (1987) evaluated the effect of feed intake level of a one-third hay, two-thirds concentrate diet on faecal digestibility of dairy cattle. Based on the intake and digestibility results reported, the decline in digestion of organic matter (OM), neutral detergent fibre (NDF) and starch was 2.9%, 6.9% and 0.1% per multiple of energy maintenance requirement, respectively. This may be related to the ratio between ruminal fractional passage rate (k_p) (which is usually assumed to increase when feed intake level is higher) and fractional degradation rate (k_d). Usually, the k_d of fibre is lower than that of non-structural carbohydrates, and fibre requires longer retention times to reach a certain degradability in the rumen. When k_p is raised in relation to elevated intake levels, the relative extent of degradation (which may in simple terms be calculated as k_d divided by the sum of k_d and k_p) of fibre is more affected than that of non-structural carbohydrates. Usually, tabulated feed values are based on the digestibility coefficients measured with sheep fed at maintenance level, and these coefficients are applied to cattle fed at higher intakes of diets necessitating a correction for the level of feed intake to be made. Feed evaluation systems differ significantly in calculations of digestion depression with increasing feed intake. The UK Feed into Milk system (Thomas, 2004) does not correct for the effect of feed intake level on digestibility. The NRC (2001) system for dairy cattle assumes a 4% reduction in digestibility per multiple of maintenance. In the Dutch NE system, a 3% reduction in digestibility per multiple of maintenance is assumed. However, in this system, an improvement of 2% in metabolizability and conversion of ME to NE is assumed at the same time, related to reduced CH_4 losses and increased propionic acid molar proportions at higher intake levels. At an average metabolizability of 0.57, the Dutch system thus adopts a net correction of 1.8% per multiple of maintenance. Although firm data on the actual correction for intake level are scarce and may depend on diet composition, the digestibility is usually adjusted for intake level. This correction reduces the theoretical advantage of dilution of maintenance requirement at increased intake levels

somewhat. This can also be seen from the difference in improvement of energetic efficiency in the dairy cattle example described previously (Fig. 2.3), where the improvement calculated with the Dutch system (correction 1.8% per multiple of maintenance) was smaller (namely 16%) than calculated with the UK Feed into Milk system (namely 19%).

The dilution of maintenance requirements in terms of metabolizable protein is less pronounced than that for energy (Fig. 2.4) for two main reasons. First, the required amount of protein for maintenance purposes relative to requirement per unit milk is generally relatively small, although this depends on what metabolic processes are actually included in 'maintenance'. In the Dutch metabolizable protein system for example, at 2.5 kg FPCM day^{-1}, the protein maintenance requirement is half that of the total metabolizable protein requirement, whereas in the net energy system the maintenance requirement for energy is half that of the total requirement at some 11 kg FPCM day^{-1}. Thus, dilution of maintenance in this protein evaluation system provides far less opportunity to improve the efficiency of conversion of metabolizable protein into milk protein compared with metabolizable energy. Second, the marginal efficiency of conversion of metabolizable protein into milk protein rapidly decreases at high production levels. The major reason for this decline in marginal efficiency is the significant intermediary metabolism of amino acids between the duodenum and the mammary gland, in particular metabolism by gut and liver tissues (Lapierre *et al.*, 2006).

Although increased production levels coincide with improved FCE, high production levels may negatively affect animal health or fertility, which in themselves may reduce FCE. The relationship between productivity and animal health or fertility is subject of much debate. For example, in their review, Ingvartsen *et al.* (2003) showed unfavourable genetic correlation between milk yield and incidence of mastitis, lameness and ketosis. However, there was no phenotypical relationship between milk yield and the risk of ketosis and lameness. Ouweltjes *et al.* (2007) performed a detailed experiment with heifers of low or high genetic merit, milked two or three times a day, and fed diets low or high in energy. They concluded that satisfactory udder health is possible with high milk production and that high production is not a major risk factor for udder health in the first lactation. From the same experiment, Beerda *et al.* (2007) concluded that signs for severe negative energy and protein balances and low body condition scores, all of which may be indicative of health risks, were not concentrated in the highest producing cows. Low-energy feed and extra milking did have an unfavourable effect on energy and protein balances, which emphasizes the possible effect of mismanagement on animal health risks. It is important to stress that it is difficult clearly to establish relationships between production and health, since there are substantial problems with confounding effects and unaccounted for biological correlations (Ingvartsen *et al.*, 2003). Overall, high production levels may not necessarily result in increased health and fertility problems, and consequently less than expected improvements in FCE, provided high-producing animals are adequately managed.

Efficiency on a Human-edible Food Basis

Increased animal productivity reduces the amount of feed and the amount of waste material (manure, CH_4) produced per unit of animal product. Much of the increase in livestock production is occurring in intensive systems (Gerber *et al.*, 2010), in part using feed produced on arable lands that could be growing food crops. Some 30–40% of cereals grown globally are used as feed for livestock (Godfray *et al.*, 2010). On a human-edible food basis, this may be considered inefficient. In the previous section, it was shown that much of the food energy in plant biomass is lost when it passes through animals. The number of people fed per hectare of cropland declines when grain is diverted through livestock into human edible food (Gill *et al.*, 2010; Godfray *et al.*, 2010). Unfortunately, cereal grains saved by reducing consumption of milk or beef does not necessarily all become available to people who do not have sufficient food (Stokstad, 2010), and neither would this reduction in grain used for livestock automatically result in more plant protein being grown (FAO, 2009). Yet given the growing world

population, it is important to avoid competition between feed for animals and food for humans.

Ruminants have a key role in converting plant resources that humans cannot (or choose not to) consume into high-quality human-edible food. By virtue of the use by ruminants of cell wall rich plant material, including grass, crop residues and other by-products of farms and industry, they do not necessarily compete directly with humans for food. In fact, ruminants augment the human food supply by exploiting inedible biomass to produce high-quality human-edible food. This ability to turn human-inedible products into human-edible products necessitates consideration of the basis to express efficiency (Baldwin, 1984; Gill *et al.*, 2010; Wilkinson, 2011). In terms of total energy or protein efficiency (efficiency of conversion of feed energy or protein to energy or protein in animal products), cattle are usually far less efficient than monogastrics (Table 2.1). Also, beef production is less efficient than milk production. In all cases, inputs exceed outputs when expressed on a total basis. However, in ruminant production systems, the efficiency when expressed on a human-edible basis (efficiency of conversion of human-edible energy or protein into energy or protein in animal products) is quite different from total efficiency. Ruminant systems are inherently more efficient when compared with non-ruminant systems on the basis of inputs and outputs edible to humans (Reynolds *et al.*, 2011). The return on human edible energy and protein inputs (Table 2.1) represents the contribution ruminants make to the total human food supply, with values of over 1 indicating that animals add to the total human food supply (Baldwin, 1984). Indeed, for dairy cattle, the human-edible efficiency in all situations considered in Table 2.1 is well over 1. For beef cattle, efficiencies on a human-edible basis are more variable, with efficiencies below 1 when diets contain a significant amount of grain. Overall, increased use of feed grains generally increases total efficiency (see previous section), but decreases returns on human edible inputs. Thus, cattle production systems based on forages and food by-products may well be net contributors of human-edible food supply, even though overall efficiency at the animal level may appear low. In contrast, cattle production systems based on high amounts of human edible products such as cereals may be net extractors of human-edible food supply, even though overall efficiency may appear high.

For dairy cattle, the above efficiency analysis is for adult cows only. The amount of energy or protein required in the period from birth to first lactation is not included in most of the analyses presented. If the amount of feed consumed during the rearing phase is included in the calculation of efficiency, both total and human edible efficiencies decline. In the Netherlands for example, the return on human edible protein input decreases from 4.38 (without rearing costs) to 3.38 (with rearing costs) (Dijkstra, unpublished results). The feed costs to grow a calf from birth until first lactation are considerable. Analogous to the effect of dilution of maintenance when production levels increase, the feed inputs for rearing are diluted when cows produce more milk and/or when an increase in longevity (the productive period per cow) is achieved (Van Vuuren and Chilibroste, 2013). For example, the return on human edible protein input including the rearing phase when culling cows after their first lactation or after six lactations is 2.04 and 3.74, respectively (Table 2.1; Dijkstra, unpublished results). Thus, dilution of feed inputs during the rearing phase increases efficiency of milk production. Management strategies to reduce involuntary culling need to be optimized to improve efficiency. However, it should be noted that milk and beef production systems in many countries are closely interlinked, as fattening of surplus calves from dairy farming and culled dairy cows provide a significant amount of beef. Feed efficiency and environmental impact of the full system (milk and beef) are not necessarily enhanced when production level or longevity of dairy cattle increases, since the improvement of efficiency on the dairy side may be party or completely offset by a decline in efficiency on the beef side (Zehetmeier *et al.*, 2012).

Finally, several studies have emphasized the role of milk and beef in adequate nourishment of humans. Animal products are relatively complete, energy-dense, rich in high-value protein, and important for the bio-available micronutrients such as vitamin B-2, vitamin B-12, vitamin D and zinc (FAO, 2011). Animal protein sources are superior to plant proteins in cereal-based diets as they provide a complete source of protein

Table 2.1. Total efficiency and human-edible efficiency[a] of different livestock production systems.

			Total efficiency		Human edible efficiency	
Reference	Country	System	Energy	Protein	Energy	Protein
Baldwin (1984)	USA	Milk	0.23	0.29	1.01	1.81
		Beef[b]	0.04	0.04	0.71	1.15
		Swine	0.23	0.38	0.58	0.86
CAST (1999)	Kenya[c]	Milk	0.07	0.09	NC[d]	NC
		Beef	0.01	0.01	NC	NC
		Swine	0.16	0.10	0.54	0.39
		Poultry meat	0.23	0.38	0.89	2.24
	South Korea	Milk	0.26	0.19	3.74	14.3
		Beef	0.06	0.06	3.34	6.57
		Swine	0.20	0.16	0.35	0.51
		Poultry meat	0.21	0.34	0.30	1.04
	USA	Milk	0.25	0.21	1.07	2.08
		Beef	0.07	0.08	0.65	1.19
		Swine	0.21	0.19	0.31	0.29
		Poultry meat	0.19	0.31	0.28	0.61
Wilkinson (2011)	UK	Milk	0.22	0.18	2.13	1.41
		Beef[e]	0.03–0.08	0.04–0.12	0.16–0.53	0.33–1.09
		Swine	0.11	0.23	0.16	0.38
		Poultry meat	0.22	0.33	0.30	0.48
Dijkstra (unpublished results)	Netherlands	Milk[f]	0.22	0.27	3.57	4.38
		Milk; average 3.5 lactations[g]	0.19	0.23	2.91	3.38
		Milk; 1 lactation	0.13	0.16	1.89	2.04
		Milk; 2 lactations	0.17	0.20	2.51	2.84
		Milk; 3 lactations	0.18	0.22	2.81	3.24
		Milk; 6 lactations	0.20	0.25	3.15	3.74

[a]Total efficiency: output of human-edible energy or protein compared with total energy or protein input with feed; human-edible efficiency: output of human-edible energy or protein compared with human-edible energy of protein input with feed. [b]Average of two beef production systems. [c]Data from Kenya presented in Table. Data from other countries presented by CAST (1999) (Argentina, Egypt and Mexico) are comparable to data from Kenya. [d]Human-edible returns for Kenya not calculated because human-edible inputs are very low or zero, which would have resulted in values approaching infinity. [e]Beef systems considered: upland suckler beef; lowland suckler beef; 18–20-month beef; ‘cereal’ beef. [f]Assuming dairy cow diet and production characteristics as presented in Bannink *et al.* (2011); proportion of human-edible energy or protein in concentrates and by-products assumed to be 0.25. [g]Including energy and protein input with feed required from birth of calf to first lactation. Feed required from birth until first lactation 4500 kg DM with proportion of milk replacer and concentrate 0.19; proportion of human-edible energy or protein in milk replacer and concentrates 0.42. In the Netherlands, average number of lactations is 3.5 (life time milk production 30,000 kg milk). Efficiency also for cows culled at first lactation (7700 kg milk), second lactation (16,200 kg milk), third lactation (25,200 kg milk) or sixth lactation (52,200 kg milk).

(i.e. containing all essential amino acids), whereas vegetable sources generally lack one or more of the essential amino acids. The nutritional value of dietary proteins can be assessed adequately by expressing the content of the first limiting essential amino acid of a protein as a percentage of the content of the same amino acid in a reference pattern of essential amino acid required by humans, corrected for true faecal digestibility of the dietary protein. This scoring method is the protein digestibility-corrected amino acid score (PDCAAS) (Schaafsma, 2000). The PDCAAS of milk is well above 100% (although values higher than 100% are truncated to 100%) and higher than many plant protein sources including peas, beans and wheat (PDCAAS between 40% and 90%). The high nutritive value of ruminant products needs to be taken into account when evaluating feed resources or excretion of waste material in various production systems. For example, Smedman *et al.* (2010) calculated the composite nutrient density of 21 essential nutrients in relation to costs in greenhouse gas (GHG) emissions from a life cycle assessment (LCA) methodology of various beverages. The GHG emission of milk production is relatively high, but the Nutrient Density to Climate Impact index (NDCI) of milk was 0.54, compared with 0.28 (orange juice) and 0.25 (soy drink). Thus, the high nutritive value of milk and beef positively impacts the efficiency of ruminant production systems when expressed on a human-edible basis and in this respect livestock may add to the food supply beyond what can be provided by crops.

Production Efficiency and Emissions

In view of the expanding world population, where prospects to increase significantly the amount of arable agricultural land are small, efficiency of ruminant production needs to increase. Or, looking from another angle, there is a need to reduce waste of natural resources in ruminant production systems. An increase in productivity (amount of product per animal) not only offers a pathway to satisfy increasing demands for milk and beef but also a possible mitigation approach to reduce the emission of various pollutants. In this chapter, we focus on the excretion of N in faeces and urine and on enteric formation of CH_4, being the major GHG in cattle production systems. Complex interactions occur between various GHG and their relation with other aspects of sustainability, such as eutrophication and animal welfare. For example, mitigation options aimed at reducing urinary N excretion may result in elevated CH_4 emission levels (Dijkstra *et al.*, 2011). The trade-off between N excretion and enteric CH_4 production needs to be understood at the animal level to allow accurate data to be used at the whole farm level. Detailed analyses using LCA methodologies (de Boer *et al.*, 2011), which recognize these interactions, are addressed in Chapter 14.

Nitrogen efficiency

Ruminants have a lower overall efficiency of N utilization and a higher excretion of N in faeces and urine per unit N intake compared with non-ruminants (Kohn *et al.*, 2005). Several factors affect this N efficiency of cattle. In general, the level of N intake itself is a major factor that determines efficiency. High dietary N levels may result in low efficiencies in both beef (Yan *et al.*, 2007) and dairy cattle (Huhtanen and Hristov, 2009). In particular, urinary N excretion increases rapidly when N intake increases (Kebreab *et al.*, 2002), which is of particular importance given that N in urine is susceptible to rapid volatilization as ammonia. Yan *et al.* (2007) observed a moderate negative relationship in beef cattle between level of feeding (as an indication of growth rate) and N excretion as a fraction of N intake, whereas a strong negative relationship between dietary N content and N efficiency was present. Similarly, in a review, Calsamiglia *et al.* (2010) concluded that high N efficiency in dairy cattle is associated in particular with low dietary N levels, but also to a smaller extent with high production levels. In dairy cattle, reduced dietary N levels may impair feed intake and/or milk production (Law *et al.*, 2009; Brun-Lafleur *et al.*, 2010). This lowered productivity may actually reduce FCE, even though N efficiency is improved. In early lactation though, it may be hypothesized that low-protein diets can improve energy status of dairy cattle by reducing milk energy output, and this may contribute to reducing the various health and fertility problems associated with severe negative energy balance and consequently improve FCE.

Nitrogen losses in the rumen occur primarily because of an imbalance between degradation of N-containing substrates and use of available N by the microbes, resulting in elevated ruminal ammonia concentrations. Key factors of efficiency in the rumen include supply of fermentable carbohydrates and the modification of protein degradation rate. The potential to recycle N within the ruminant is a key issue in reducing N losses from the rumen. Recycling of urea-N to the gut is significant. Lapierre *et al.* (2005) reported 0.09–0.81 of hepatic ureagenesis to return to the gut, equivalent to 0.04–0.73 of the digested N, via the portal-drained viscera (PDV). This recycling is of particular importance with low-protein diets where fractional urea re-absorption from kidney and urea clearance rate increase. Efficiency of microbial protein synthesis in the rumen is also positively related to fractional passage rate, since high fractional passage rates from the rumen result in high fractional growth rates of microbes, and the principle of dilution of maintenance discussed previously for energy and protein utilization by ruminants is also valid for microbial nutrient utilization and will result in elevated microbial efficiencies (reviewed by Dijkstra *et al.*, 2007). In general, higher feed intake levels will increase fractional passage rates from the rumen and therefore may improve microbial efficiency and efficiency of N utilization in the rumen.

Metabolism of absorbed amino acids in the PDV and the liver is considerable. Splanchnic tissues have a high metabolic activity. Splanchnic tissue protein contributes only 6–9% to total body protein (Gibb *et al.*, 1992), but accounts for more than 50% of whole body protein synthesis (Lapierre *et al.*, 2005). On average, 35% of amino acids are lost during absorption and the liver removes some 45% of absorbed amino acids (Lapierre *et al.*, 2005). Among the major factors affecting post-ruminal N efficiency is the amount of NE available (Doepel *et al.*, 2004), partly mediated by the effect of energy on rumen microbial protein synthesis as shown in the simulation study by Kebreab *et al.* (2002). Further opportunities to decrease N losses post-ruminally arise from proper balancing of diets for individual amino acids. The difference between metabolizable protein allowable milk yield and actual milk yield increases as the concentration of lysine or methionine decreases below a threshold value (NRC, 2001). From a comprehensive review on the effects of balancing dairy cattle diets for individual amino acids, Schwab *et al.* (2005) concluded that improving the balance of individual amino acids, in particular lysine and methionine, will increase N efficiency. Quantitative understanding of the utilization of amino acids by organs and tissues is still largely lacking and improved understanding may allow further development of strategies to improve N efficiency in ruminants (Calsamiglia *et al.*, 2010).

Methane efficiency

Enteric CH_4 production arises principally from microbial fermentation of hydrolysed dietary carbohydrates and (quantitatively less important) protein, in which volatile fatty acids (VFA), CO_2 and hydrogen (H_2) are formed. Hydrogen and CO_2 are the primary substrates for methanogenesis in ruminants. Methane is formed by methanogenic archaea (methanogens). About 90% of CH_4 is formed in the rumen, and the remainder in the hindgut. Enteric CH_4 production varies widely and ranges from 2% to 12% of gross energy (GE) intake (Johnson and Johnson, 1995). Level of feed intake and composition of the diet are the main factors affecting CH_4 formation in the ruminant. Increased intake levels are associated with proportionally less CH_4 production (Johnson and Johnson, 1995; Mills *et al.*, 2001), because higher intake levels are associated with reduced rumen retention times, reduced rumen pH and a consequent shift towards propionic acid formation (Ellis *et al.*, 2008). Thus, elevated production levels associated with increased feed intake levels not only reduce CH_4 per unit product due to the dilution of maintenance effect, but also reduce CH_4 per unit product due to a decreased proportion of feed converted to CH_4. With respect to diet composition, fermentation of structural carbohydrates leads to more CH_4 produced per unit feed digested than fermentation of starch and non-structural carbohydrates (Bannink *et al.*, 2008). In this respect, CH_4 production by cattle is a necessary consequence of transforming human-inedible feed

(fibre-rich forages and by-products) into human edible food. Cattle production systems based on fibrous feeds may be associated with high levels of GHG emissions per unit product, prompting general calls to reduce the consumption of beef and milk products, but in many situations where the output of human-edible food is larger than its input, cattle production systems still have a major role in food security.

There are several other dietary options to reduce enteric CH_4 formation that have been discussed in various reviews (Ellis *et al.*, 2008; Iqbal *et al.*, 2008; Beauchemin *et al.*, 2009; Martin *et al.*, 2010; Cottle *et al.*, 2011; Grainger and Beauchemin, 2011). The addition of dietary fat or oil reduces CH_4 production by about 5% per 1% added lipid to the diet (reviewed by Grainger and Beauchemin, 2011). Dietary lipid supplementation decreases CH_4 production mainly by reducing the activity of methanogens and protozoal numbers, and reducing fermentation of substrate in the rumen. This CH_4 reduction appears not to be influenced by the type of fatty acid (saturated versus unsaturated; carbon chain length) as long as the addition of fat does not negatively affect feed intake or nutrient digestibility (Van Zijderveld *et al.*, 2011a). Rumen modifiers such as monensin may efficiently reduce CH_4 production, but the persistency of the effect has been questioned (Martin *et al.*, 2010). Yeast cultures based on *Saccharomyces cerevisiae* are widely used on commercial dairy farms in North America and Europe to improve milk yield and production efficiency, and thus may decrease CH_4 per unit product, although other effects including a shift in partitioning of nutrients to microbial biomass and various VFA are not documented well (Grainger and Beauchemin, 2011). Alternative electron acceptors, in particular nitrate, may be highly effective in reducing CH_4, but require careful introduction and adaptation of cattle in view of the risk of nitrite poisoning (Nolan *et al.*, 2010). In dairy cattle, nitrate has been shown to have a persistent CH_4 reducing effect for at least 4 months (Van Zijderveld *et al.*, 2011b), corresponding to its mode of action. In contrast, a number of dietary additives that have shown promising results *in vitro* have not been effective *in vivo* (Reynolds *et al.*, 2011; Van Zijderveld *et al.*, 2011a).

Production Level, Nitrogen Efficiency and Methane Production

As discussed previously, increased production is a powerful means of reducing waste excretion per unit product. Potential improvements in N and CH_4 efficiency are most pronounced at low levels of production. At higher production levels, according to the law of diminishing returns, efficiency gains are smaller, but still worthwhile to pursue. An example of efficiency gains for N and for CH_4 production per unit product is presented in Fig. 2.5. Annual milk production (fat and protein corrected milk, FPCM) in the Netherlands increased from 6270 (1990) to 8350 (2009) kg per cow, with a rise in feed intake of about 18%. The FCE (kg feed DM per kg FPCM) decreased from 0.88 to 0.78. The N efficiency (g N in milk g^{-1} N intake with feed) increased from 0.18 to 0.27, and CH_4 declined from 17.6 to 15.4 g kg^{-1} FPCM, when comparing 1990 and 2009 (Bannink *et al.*, 2011). Similar improvements in efficiency and reduction of waste per unit product have been shown in other dairy and beef cattle production systems (e.g. Capper *et al.*, 2009; Capper, 2011). The rise in N efficiency was due in particular to a decline in dietary N-content, whereas the reduced CH_4 production per unit milk was related particularly to the increased feed intake level. Such improvements were achieved without increasing the proportion of concentrates and wet by-products in the diet (which actually decreased very slightly, from 0.30 to 0.29 of diet DM) whilst reducing N fertilizer input. Thus, major gains in reduction of excreta and emissions from cattle production systems are possible.

It is crucial to note that results based on comparisons between systems are not necessarily a good reflection of the results achieved when improving productivity within a system. Gerber *et al.* (2011) explored the relationship between productivity of dairy production and GHG emissions on a global scale. A LCA methodology was used that included the emissions of CO_2, CH_4 and N_2O, for 155 countries. Gerber *et al.* (2011) presented two equations to predict GHG emissions per kg FPCM (Fig. 2.6). These equations, based on the same dataset, have different implications. In the non-linear equation they derived, GHG emissions per kg FPCM stabilize at a milk production level of about 6000 kg $year^{-1}$ and the asymptotic value of emissions is 1.4 kg CO_2-e $year^{-1}$. In the

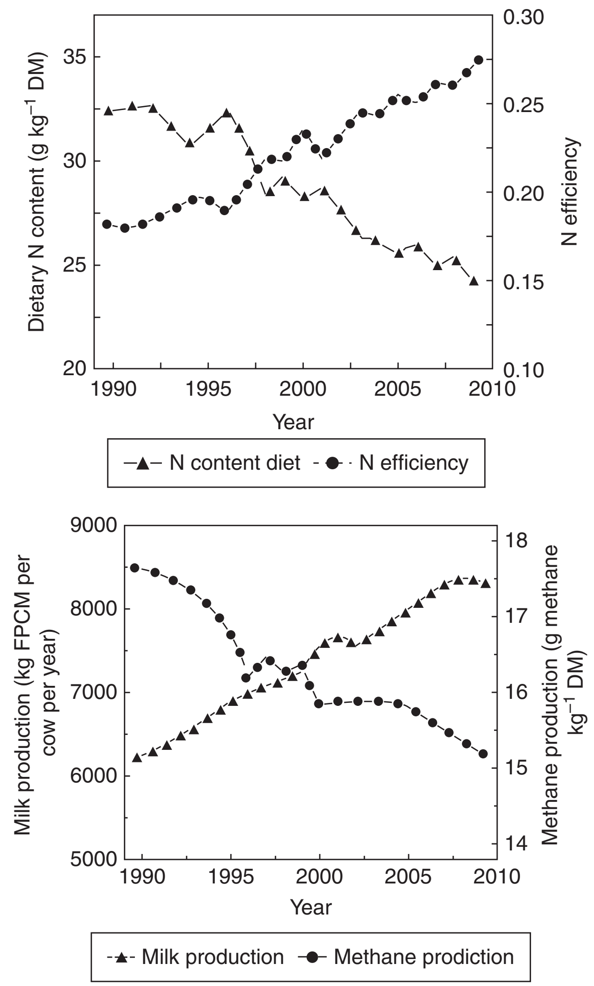

Fig. 2.5. Dietary nitrogen (N) content (g kg^{-1} DM) and N efficiency (g milk N g^{-1} feed N) (top) and milk production (kg fat and protein corrected milk (FPCM) per cow per year) and methane (CH_4) production (g CH_4 kg^{-1} FPCM) (bottom) for the average dairy cow in the Netherlands from 1990 until 2009. Data and CH_4 calculations from 1990 until 2008 as described in Bannink *et al.* (2011) with 2009 results added according to the various sources and calculation procedures mentioned by Bannink *et al.* (2011).

linear equation though, further reductions in GHG emissions per kg FPCM are possible with an asymptotic value of 0.86 kg CO_2-e year^{-1}. In this analysis of dairy production systems with one average value per country in the database, increased milk production reduced emissions of CH_4 and N_2O per kg milk, but increased CO_2 emission per kg milk which according to Gerber *et al.* (2011) reflects the increased use of inputs whose production requires fossil fuel, notably high-value feed. In deriving the linear and non-linear equations, Gerber *et al.* (2011) used different responses (CO_2-e per cow per year, or per kg FPCM). It is difficult to discriminate on statistical grounds between the two equations. Problems with model assumptions on normality and homoscedasticity may occur with both equations. Capper *et al.* (2009) reported a decline in GHG emission of 3.6 kg CO_2-e kg^{-1} FPCM (1944; 2061 kg FPCM per cow per year) to 1.35 kg CO_2-e kg^{-1} FPCM (2007; 8715 kg FPCM per cow per year). According to the non-linear equation of Gerber *et al.* (2011), the GHG emission would be 2.51 and 1.38 kg CO_2-e kg^{-1} FPCM, respectively. The linear equation

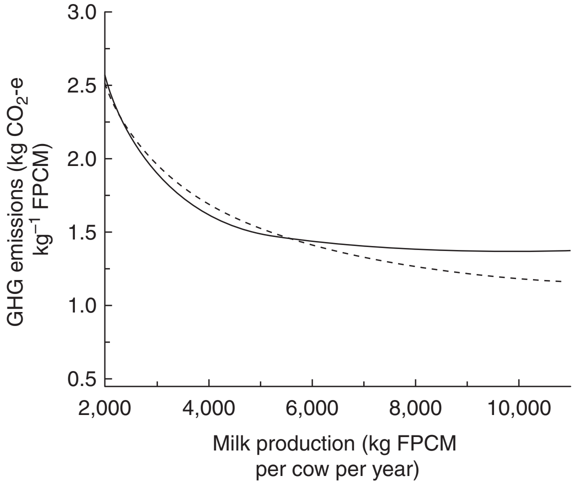

Fig. 2.6. Relationship between milk production level (kg fat and protein corrected milk (FPCM) per cow per year) and total greenhouse gas emissions (GHG; expressed in kg of CO_2 equivalents per kg fat and protein corrected milk, FPCM). Prediction equations based on a database containing 155 countries described by Gerber *et al.* (2011). Solid line: $GHG=1.373552\times\exp[2.3837\times(0.999332^{FPCM})]$; dashed line: $GHG=3315.5/FPCM+0.8649$.

predicts a somewhat more pronounced decline (2.47 and 1.25 kg CO_2-e kg^{-1} FPCM, respectively). For the dairy production system in the Netherlands (Fig. 2.5), the non-linear equation would predict just a 2% decline in GHG emission (from 1.42 down to 1.39 kg CO_2-e kg^{-1} FPCM in 1990 and 2009, respectively), whereas the linear equation predicts a 10% decline (from 1.39 to 1.25 kg CO_2-e kg^{-1} FPCM in 1990 and 2009, respectively). However, the reduction in CH_4 per kg FPCM in the Dutch dairy system is in itself already 0.06 kg CO_2-e kg^{-1} FPCM. Since the N content of the diet was reduced from 1990 to 2009, and given that FCE improved by more than 10%, which will also decrease the emissions of N_2O and CO_2, the estimate of the non-linear equation understates the reduction in GHG emission upon increased production levels under the Dutch system. Clearly, the analysis based on between-country data should not be used for within-country calculations of productivity increases.

Conclusions

Major gains in efficiency of ruminant production systems are needed. With no prospects to increase the amount of arable agricultural land significantly, food production must intensify to ensure an affordable, abundant food supply. In particular, the ability of ruminants to turn human-inedible products into human-edible products will become increasingly important in terms of global food security. Increases in ruminant production will largely have to come from further advancement in the efficiency of livestock systems in converting natural resources into human-edible food. Intensive production systems are a source of concern due to environmental impacts including N pollution and GHG emissions. Past improvements demonstrate the ability of production efficiency to decrease the environmental impact per unit of milk, particularly that of enteric CH_4 emissions. Improvements in feed digestibility and better post-absorptive matching of absorbed nutrients with production level must be achieved, whilst at the same time not increasing the human-edible inputs per unit livestock product. In improving the efficiency of nutrient use for milk and beef production by cattle, and their environmental impact, it is imperative that the wider animal health and welfare, social and economic implications of policies to increase productivity are considered.

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3 Production Efficiency of Monogastric Animals

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Introduction

The number of farm animals has enormously increased over the last few decades globally to meet the increased demands for animal products caused by the expanding world population (FAO, 2011). It is forecasted that the demand for meat will increase by more than 70% along with the increase in population size to more than 9 billion in 2050 (United Nations Population Division, 2010). This is expected to cause a more intensified animal husbandry as the arable land area will not increase accordingly.

Over the last 100 years, land use has changed dramatically as the percentage of total land area used for cropland and pasture has expanded at the expense of natural habitats (Steinfeld *et al.*, 2006). The development in production of monogastric animals is closely related to these changes in land use as it relies on the production of cereals and protein crops (Steinfeld *et al.*, 2006). Since 1960, the production of soybeans and cereals has increased three- to tenfold concomitantly with an increase in soybean harvest area of about threefold and a stagnation in cereal harvest area (Steinfeld *et al.*, 2006). The increased crop production is therefore mainly caused by an increased cropping intensity and yield partly due to increased access to inorganic fertilizers (Erisman *et al.*, 2011). According to Steinfeld *et al.* (2006), feed production is estimated to use about one-third of all cropland, and the demand for animal feed will increase along with the expansion in the number of livestock. In the past, livestock production was directly related to access to arable land, but the industrialized animal production has in many cases been decoupled from crop production at a local or even regional scale. This is challenging to the modern industrialized animal production and calls for improvements in production efficiency in order to improve the utilization of nutrients and to lower the impact of animal production on the environment.

This chapter deals with ways to improve the production efficiency and reduce the excretion of primarily nitrogen (N) and phosphorus (P), but also other undesired substances such as heavy metals in monogastric animals and emissions of ammonia and methane. Two main areas will be addressed: (i) the amount of feed required; and (ii) the concentration and quality of nutrients in the provided feed. The first topic focuses on how much feed the animal requires for the

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production of 1 kg of meat or eggs defined as the feed conversion ratio (FCR). The second topic covers the amounts of nutrients and energy fed to the animal in relation to the feed, and the digestibility and availability of the nutrients.

Feeding and Feedstuffs in Monogastric Animals

Monogastric animals are normally fed cereal-based diets supplemented with vegetable protein sources such as soybean meal, canola and other de-oiled by-products. In the past, it was the norm to include feedstuffs of animal origin, but due to restrictions and costs, the use of these feedstuffs is often limited, but may depend on regional concern and restrictions. The diets are typically supplemented with minerals and vitamins and other feed additives.

The digestive processes taking place are basically the same in pigs and poultry. They are true omnivores, but under farm conditions, a large proportion of the feed is of plant origin. The digestive processes mainly take place in the upper part of the gastrointestinal tract, but fibrous materials are partly fermented and utilized in the large intestine. The small intestine is the main site for absorption of nutrients, but some nutrients and fluids are also absorbed in the large intestine (Lærke and Hedemann, 2012).

The animals must be provided sufficient amounts of feed to ensure health and production. Compound feed is composed of different feedstuffs and supplements by use of feeding programmes enabling the industry to formulate feed that contains sufficient amounts of energy and nutrients to sustain the animals' performance and health and the quality of the end-products, meat or eggs. Computerized programmes for diet optimization are often based on mathematical models that estimate the sufficient amounts of energy, amino acids and usually also certain minerals such as phosphorus (P) and calcium. These amounts are adapted to the requirement set by the different species at specific body weights or level of performance. The models are continuously refined and updated along with the development of new knowledge and practical experience. Individual countries (or regions) often have their own programmes modified to local conditions or established preconditions, but the fundamental principles are largely the same (e.g. NRC, 1994, 1998, 2012; Pig Research Centre, 2012). Increasingly these feed optimizing programmes also include 'a green module' where the output of nutrients and emissions is estimated enabling the user to include an environmental aspect in his feed optimization. Different models and equations have been widely used to quantify the nutrient excretion and efficiency for a couple of decades (e.g. Dourmad *et al.*, 1992; van der Peet-Schwering *et al.*, 1999; Poulsen *et al.*, 2006; Kebreab *et al.*, 2008).

Conversion of Feed into Edible Products

The feed supply should cover the animal's physiological need for nutrients and energy for production and for the maintenance processes in the body. Maintenance requirements cover the amounts that are needed to keep up the basic functions of the organs and body tissues and comprise mainly energy and protein but occasionally also some minerals. Generally, maintenance requirements are given first priority by the animal and growth and production are the second priority.

The animal's physiological need for energy and nutrients has to be met by intake of sufficient amounts of feed, but the intake should be tailored to fulfil the animal's minimum requirement to avoid excessive supply and unnecessary excretion of nutrients. Feed intake is regulated by complex processes that involve both metabolic and physical mechanisms. These mechanisms comprise a short-term process related to the fed meal and a more long-term regulation. Growing animals mainly rely on the short-term regulation, whereas the long-term regulation is more related to reproducing adult animals. This complex regulation of feed intake has recently been reviewed by Torrallardona and Roura (2009).

Part of the feed consumed is used for maintenance and concerns both energy and essentials nutrients. The thermal environment in which the animal is kept influences the feed intake, and animals kept in cold environments consume more feed as they need more energy for

thermal well-being, whereas animals kept in thermo-neutral conditions have no extra need for energy as the heat production is at a minimum. In hot areas, the animal's capacity to get rid of the heat generated by the metabolic processes taking place within the body often reduces feed intake (Tauson, 2012). Consequently, the climate or housing conditions may have substantial impacts on feed intake affecting the efficacy of turning feed into edible products. Therefore, the concentration of nutrients (protein, amino acids, minerals etc.) has to be adjusted to the energy supply in order to optimize the intake to the animal's requirement. The actual requirement for energy and nutrients is dependent on the physiological stage of the animal and therefore the use of phase feeding where the feed composition is adjusted to the changes in the nutrient and energy requirement is widely practised (Kebreab *et al.*, 2012). This means that several feed mixtures are used to raise a broiler or a growing finishing pig.

Genetic selection programmes have led to a substantial increase in the capacity for lean tissue growth and decrease in the rate of fat deposition in monogastric animals (Jensen *et al.*, 2011). At the same time, the selection for lean tissue growth (less deposition of fat) has also improved average daily gain (ADG) and FCR, as the feed amount needed to produce 1 kg lean meat is much lower compared with the need for producing 1 kg fat. A major tool to improve the utilization of all nutrients and energy is therefore to minimize the amount of feed needed to produce 1 kg of product. The goal is to reduce both the maintenance requirements and the number of days required to produce a pig or a chicken to a minimum. The overall principles of nutrient utilization are shown in Fig. 3.1.

Selection programmes for higher ADG and lower FCR are widely applied (Jensen *et al.*, 2011). Animals have undergone selection for productivity, which means that modern breeds demand less feed for producing a unit of product. Actually, the FCR is often as low as about 1.5 in chicken (during the 30 days to reach a body weight of 1.6 kg) and 2.6 in growing finishing pigs (30–110 kg body weight) (Poulsen *et al.*, 2012). All things being equal, an improvement in FCR has major impact on the general utilization of energy and nutrients. The development in FCR in growing finishing pigs is shown in Fig. 3.2.

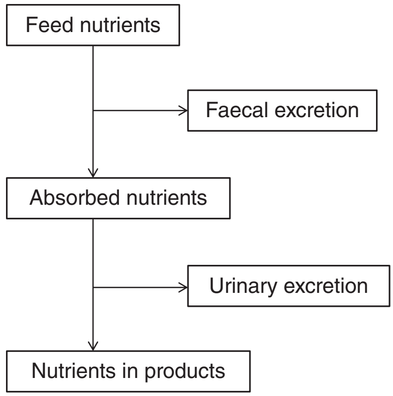

Fig. 3.1. Conversion of nutrients in feed into products and excretions.

As it appears later in this chapter, the dramatic improvements during the 1980s and 1990s are partly due to the more efficient protein utilization and subsequent growth.

Growth is usually expressed in daily gain but can also be expressed in protein retention, which emphasizes that focus has to be put on protein accretion to obtain lean growth. All things being equal, this will also result in the most efficient conversion of feed into edible products for human consumption.

Protein, Amino Acids and Nitrogen Utilization and Efficiency

Protein is an essential dietary component for all animals, but it is not protein *per se* but the amino acids as constituents of proteins that play a role (Lewis and Southern, 2001). Basic knowledge of the importance of amino acids in relation to productivity (growth and reproduction) has been critical for improvements in N utilization in monogastric animals (Jensen *et al.*, 2011). It is recognized that some of the 20 different amino acids present in proteins have to be supplied by the feed every day and are thus considered essential amino acids, whereas others are non-essential, i.e. they do not need to be provided as they can be synthesized by the animals (Table 3.1). Three of the amino acids are considered semi-essential as they can only be synthesized from essential amino acids (cysteine from methionine (both contain sulfur) and tyrosine from

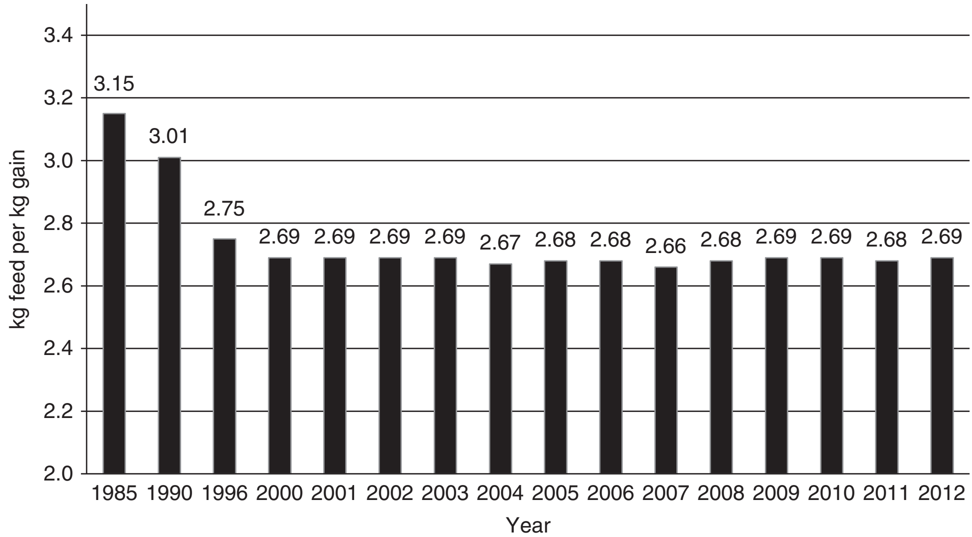

Fig. 3.2. Development in feed conversion rate (FCR) in Danish grower-finisher pigs, kg feed per kg gain from 1985 to 2012. The initial body weight is 30 kg during all years, but the body weight at slaughter has increased from 95 to 107 kg in 2012.

Table 3.1. Alphabetical classification of amino acids according to their essentiality

Essential amino acids	Non-essential amino acids	Semi-essential amino acids
Histidine	Alanine	Arginine
Isoleucine	Asparagine	Cysteine
Leucine	Aspartic acid	Tyrosine
Lysine	Glutamic acid	
Methionine	Glutamine	
Phenylalanine	Glycine	
Threonine	Proline	
Tryptophan	Serine	
Valine		

phenylalanine (both containing an aromatic group). The primary limiting amino acids are lysine, methionine, threonine, tryptophan etc., but the order of limitation depends on the available feedstuffs used to compose the feed for the animals. There may also be species-dependent differences as poultry in general has a high requirement for the sulfur-containing amino acids for feather production.

In the gastrointestinal tract, the protein fed to the animals is subjected to luminal digestion releasing free amino acids and di- and tripeptides that are absorbed and transported into the circulating blood (Lærke and Hedemann, 2012). The fed amounts of crude protein that are digested are called digestible protein. The protein digestibility depends on the composition of the feed and varies between species (and categories) and among different ages. Often, the protein digestibility is above 80% in pig and poultry diets, but when large amounts of e.g. fibrous feeds are included, the protein digestibility is lower. In general, the amounts that are not digested are excreted in faeces, and those that are digested and absorbed may be utilized for the synthesis of protein used for many purposes including protein accretion in the muscles or as a source of energy. If the absorbed amino acids are not used for protein synthesis, they are deaminated and the surplus

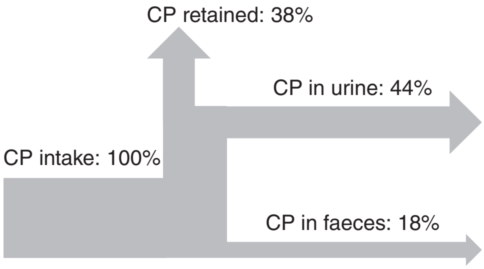

Fig. 3.3. Retained protein and excreted (faecal and urinary) nitrogen, percentage of intake in growing-finishing pigs.

N is excreted in urine (Jensen *et al.*, 2011). The overall principles of N utilization are shown in Fig. 3.3.

Protein or its constituent essential amino acids must be supplied on a daily basis in amounts equivalent to the animal's requirement. The need for protein/amino acids therefore varies during the animal's life with the highest percentage of dietary content in early life. During reproduction, the percentage of protein and amino acids is higher in diets fed to lactating sows compared with diets for pregnant sows.

Historically, the first challenge in pig and poultry production was to feed protein-sufficient diets to the animals (protein balanced feed), which resulted in increased production and N efficiency because the maintenance requirement was lowered due to the higher daily performance (Jensen *et al.*, 2011). Nowadays, focus is on feeding the animals a diet that is balanced according their amino acid requirement. All feedstuffs contain crude protein but the concentration varies (Fig. 3.4). Unfortunately, cereals and protein feedstuffs not only contain essential amino acids but also an overload of non-essential amino acids in relation to the animals' requirements. Especially, the protein in cereals contains large proportions of the amino acid glutamine, which is a non-essential amino acid. The contents of essential amino acids also differ between feedstuffs, illustrated in Fig. 3.5 by the concentrations of lysine and valine. One way to improve the N efficiency and reduce the excretion of N is to formulate diets consisting of low levels of protein supplemented with essential amino acids in amounts sufficient to cover the animal's amino acid requirement (Nørgaard, 2012).

Fortunately, the occurrence of industrially produced amino acids (crystalline amino acids) on the market gave rise to reductions in crude protein content by inclusion of crystalline amino acids. Importantly, the dietary crude protein can, concurrently with the supplementation of crystalline amino acids, be lowered by less inclusion of e.g. soy protein (Fig. 3.6).

This combined action has been the major feeding measure contributing to the lowered N excretion that has been observed over the last decades (Fig. 3.7). From 1985 until 2012, the excretion of N has been reduced 51% in grower-finisher pigs.

The first commercially relevant crystalline amino acid was lysine, but now other amino acids such as methionine, threonine, tryptophan and valine are widely used in the feeding of monogastric animals (Fig. 3.6). The use of crystalline amino acids depends largely on the need to reduce N excretion, which especially in livestock-dense regions or countries is of major concern, because in those areas, regions or countries strict demands have been defined politically. For instance, the European Community (EC) introduced directives to alleviate the risk of leaching of nitrate to prevent pollution of groundwater and soft water ecosystems 20 years ago (Oenema *et al.*, 2011). However, the costs for the replacement of soybean meal, canola, rapeseed etc. with the crystalline amino acids greatly affect to what extent these amino acids are used in practical feeding situations.

The actual utilization and excretion of N in poultry is shown in Table 3.2. The utilization of protein is high in broilers whereby the N excretion is less than 50% of the intake, but the utilization is highest in the youngest broilers and decreases with age. Broilers raised organically have a much lower protein efficiency compared with conventionally raised broilers, whereby the N excretion is much greater in organically raised birds. Laying hens have a high excretion and low utilization of N compared with broilers, with the poorest efficiency in organically raised laying hens (Table 3.2).

Recently, political focus has also been on the emissions of ammonia in agriculture. This also calls for actions in livestock farming, especially in areas with intensive animal production. Therefore, restrictions on manure application and on-farm handling of manure have been

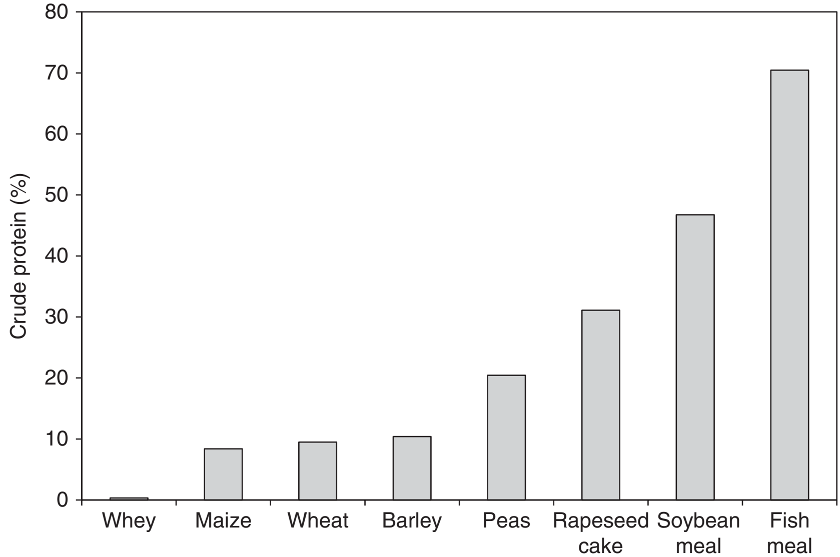

Fig. 3.4. Crude protein concentration, percentage in commonly used feedstuffs (as-fed basis) (Nørgaard, 2012).

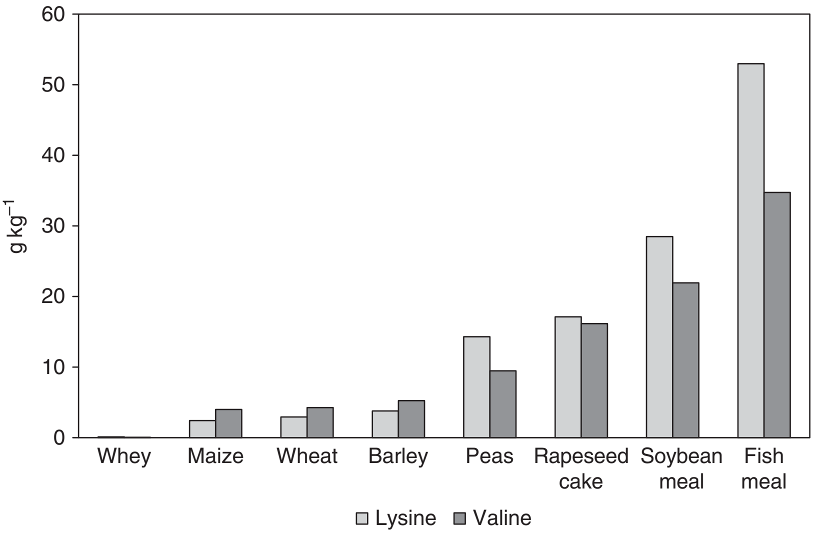

Fig. 3.5. Examples of amino acid concentrations in commonly used feedstuffs, g kg^{-1} (as-fed basis) (Nørgaard, 2012).

introduced in e.g. the EC. Local demands for specific low-ammonia housing systems for livestock have been established. However, all things being equal, the focus on reductions in N excretions by feeding means has also decreased the emission of ammonia. Ammonia emission (housing and manure storage facilities) from the production of monogastric animals has been covered in many papers and reviews (e.g. Monteny and Hartung, 2007) and will not be discussed further.

In summary, two major factors are the driving forces behind the improvement in N use in

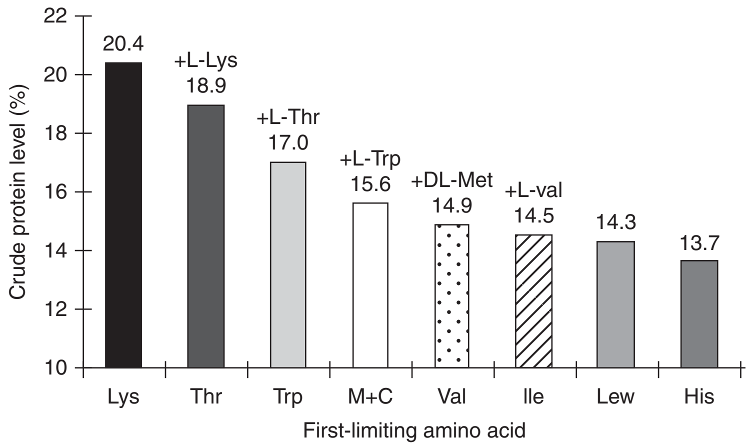

Fig. 3.6. Substitution of crude protein by amino acids (Nørgaard, 2012).

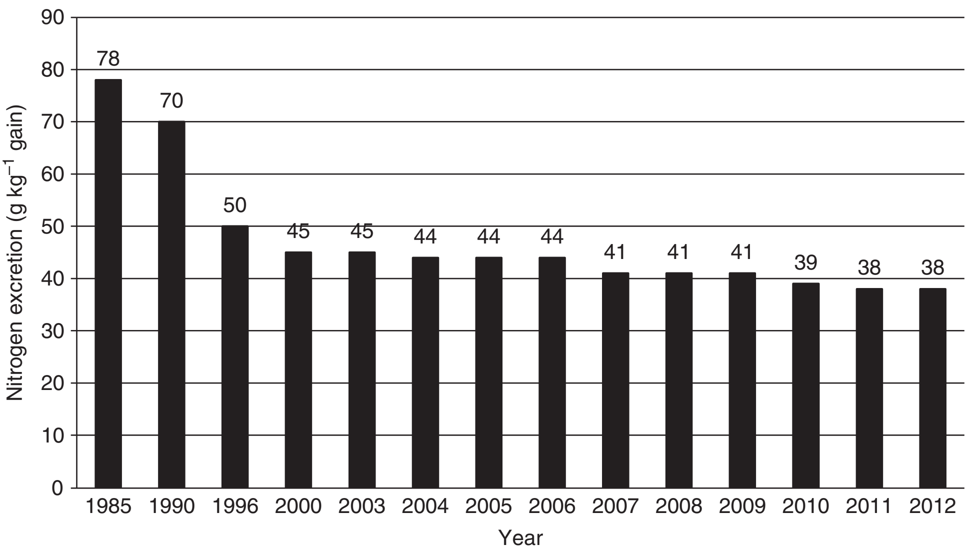

Fig. 3.7. Nitrogen excretion from grower-finisher pigs, g kg^{-1} gain from 1985 to 2012. The initial body weight is 30 kg during all years but the body weight at slaughter has increased from 95 to 107 kg in 2012.

monogastric animals, the genetic selection of animals for a more efficient production and the replacement of crude protein by industrially produced amino acids. The establishment of computerized models used to optimize the composition of the feed based on amino acid requirements is another important tool enabling the information and knowledge from research and development to be combined and used in practical feeding. It is important to remember that animals will not be effective if they are not healthy and supplied with the necessary amounts of energy and nutrients. This calls for a balanced sustainable approach involving all significant aspects of animal production so that reductions in excretion of e.g. N should not cause reductions in productivity, health and welfare (Blaha and Köfer, 2009).

Table 3.2. Nitrogen excretion and utilization in poultry (Poulsen *et al.*, 2012).

Type of poultry	Duration	Nitrogen excretion (kg)	Nitrogen utilization (% of intake)
1000 broilers	30 days	34.0	58
1000 broilers	32 days	39.2	57
1000 broilers	35 days	47.7	55
1000 broilers	40 days	64.8	53
1000 broilers	81 days, organic	126.6	33
100 female turkeys[a]	112 days	48.1	37
100 geese[a]	91 days	56.1	22
100 ducks[a]	52 days	17.3	34
100 layers hens, batteries	1 year	69.3	34
100 laying hens, free living	1 year	82.9	29
100 laying hens, organic	1 year	101.1	25

[a]The production of these birds is limited in Denmark and the calculated excretions are only indicative.

Phosphorus Utilization and Efficiency

Many minerals are essential dietary components for all animals but they are only required in very small amounts, macro-minerals in grams per kilogram and trace minerals in parts per million (mg kg^{-1}) or less. The functions of the minerals are diverse and range from structural functions in the body to regulatory components of substances and enzymes in tissue and blood. Phosphorus interacts with calcium and consequently plays a major role in the development and maintenance of the skeleton in all animals. Moreover, P is substantial in many other processes such as the metabolism of energy and cell replication. It is well known that animals fed P-deficient diets develop a soft skeleton that is incapable of keeping the animals healthy and in a standing position. Therefore, for decades, it has been common practice to add inorganic minerals to diets consisting of cereals and protein sources of plant origin. Previously, bone meal or meat and bone meal was also widely used, but due to restrictions in the use of animal by-products this use is not currently allowed in some areas (e.g. within EC). Thus, inorganic P sources are still commonly used to ensure a sufficient dietary P content to cover the animals' P need.

Most P in cereals and seeds is present as phytate, the salt of phytic acid that consists of a ring of six carbon atoms (*myo*-inositol) to which six phosphate groups are attached (Erdman, 1979). Phytate is referred to as *myo*-hexakisphosphate or $InsP_6$ and the presence in different feedstuffs is shown in Table 3.3. Phytic acid is able to form complexes with minerals (as shown in Fig. 3.8) but also with proteins and other components present in seeds. $InsP_6$ is mainly deposited in the protein bodies of the aleurone or cotyledons in cereals and seeds. During germination and early growth, $InsP_6$ is degraded and phosphate molecules are gradually released turning the $InsP_6$ into $InsP_5$, $InsP_4$ and so forth. Unfortunately, monogastric animals do not possess sufficient capability to digest phytate whereby phosphate could be released in a similar way. Therefore, feed for monogastric animals has, as mentioned, been supplemented with phosphate of non-vegetable origin (meat and bone meal, bone meal or inorganic phosphates from mines).

Since the early 1990s, when the first reports on the effects of supplementing microbial phytase to feeds for monogastric animals were introduced (Simons *et al.*, 1990), the use of the enzyme has expanded globally. The reason for this is that the degradation of phytate is stimulated by phytase whereby the release of phosphate molecules is increased. This results in increased P absorption in pigs and poultry when feed contains phytase. For more than 20 years, the effects of phytase have been studied in hundreds of experiments with pigs and poultry. At the same time, different companies have developed different phytases that are able to dephosphorylate $InsP_6$. Comprehensive reviews have been presented (Jongbloed, 2012) or published recently (e.g. Selle and Ravindran, 2007, 2008).

Table 3.3. Phytate concentration (g kg^{-1}) and phytase activity[a] (FTU kg^{-1}) in commonly used feedstuffs (Eeckhout and De Paepe, 1994).

Feedstuff	Total P	Phytate P	Phytate P (% of total P)	Phytase activity
Wheat	3.3	2.2	67	1193
Barley	3.7	2.2	60	582
Triticale	3.7	2.5	67	1688
Rye	3.6	2.2	61	5130
Oat	3.6	2.1	59	42
Maize	2.8	1.9	68	15
Wheat bran	11.6	9.7	84	2957
Soybean meal	6.6	3.5	53	40
Rapeseed meal	11.2	4.0	36	16

[a]One phytase unit (FTU) corresponds to the amount of enzyme that liberates 1 μmol inorganic orthophosphate from 0.0051 mol l^{-1} sodium phytate within 1 min at pH 5.5 and 37°C (Engelen *et al.*, 1994).

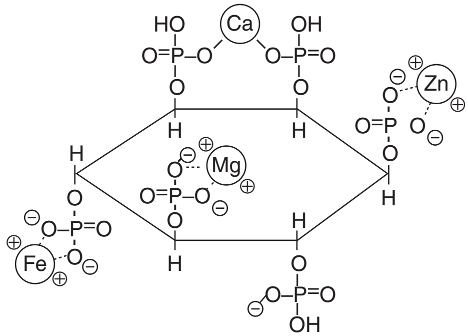

Fig. 3.8. The phytate molecule (Ins P_6; myo-hexakisphosphate; Erdman, 1979).

The efficiency of phytase on the P digestibility depends on the composition of the feed because some cereals such as maize and heat-treated processed oilseeds such as soybean meal do not possess plant phytase themselves, which is why the effects of microbial phytase supplementation are up to 0.8 g of digestible P per kg (Jongbloed, 2012). In contrast, the effect of microbial phytase addition may be about 50% less in feed containing wheat and barley, as these cereals possess plant phytase. Table 3.3 shows the plant phytase activities in commonly used feedstuffs. However, when cereals containing plant phytase activity are heat treated, phytase may be partly inactivated affecting the digestibility of phytate P. As such, it is very important to be aware of the specific feeding conditions when the digestibility of P is forecasted. Currently, only a few feed optimization programmes include appropriate preconditions to predict the accurate content of digestible P in feed for pigs and poultry.

Similar to the replacement of crude protein by crystalline amino acids, inorganic mineral phosphates (feed phosphates) have to be replaced by the addition of microbial phytase in order to improve the P efficiency. However, the exchange rate of mineral phosphates has to be adapted to the expected efficacy of microbial phytase in the feed. This understanding is important in order to avoid a too low P supply (resulting in P deficiency) and at the same time a surplus in P supply resulting in excessive P excretion. Normally, the urinary P excretion is very low but if the intake of digestible P is in surplus of the animal's requirement, the surplus is eliminated via urine. This combined action of improved knowledge on digestible P requirement, improved FCR and exchange of feed phosphate by microbial phytase has gradually lowered the P excretion in grower-finisher pigs (Fig. 3.9). From 1985 until 2012, the P excretion per kg gain has been reduced 51%. The biggest decrease was observed in the mid-1990s and reflected the change in P recommendation from total to digestible P and the onset of microbial phytase supplementation. Recently, an increase in P excretion has been observed due to a slight increase in P recommendations.

Figure 3.10 shows the relationship between P intake, P absorption, P retention and P excretion in faeces and in urine in the grower-finisher pig. At low P supply, the retention is low and the urinary excretion is also very low reflecting the inevitable loss. Increased P intake results in increased P absorption and retention, and at a certain P intake, the P retention reaches a plateau. At the same time, the urinary P excretion

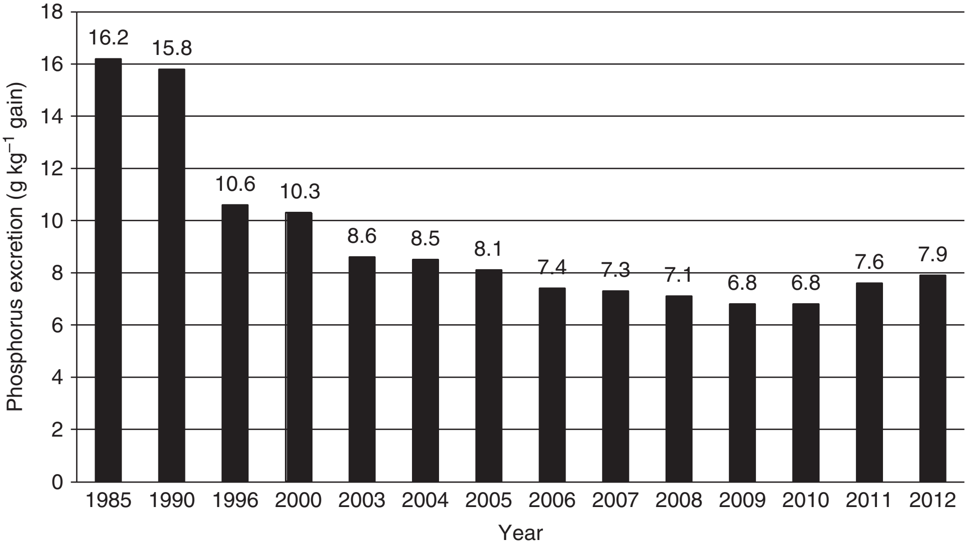

Fig. 3.9. Phosphorus excretion from grower-finisher pigs, g kg^{-1} gain from 1985 to 2012. The initial body weight is 30 kg during all years, but the body weight at slaughter has increased from 95 to 107 kg in 2012.

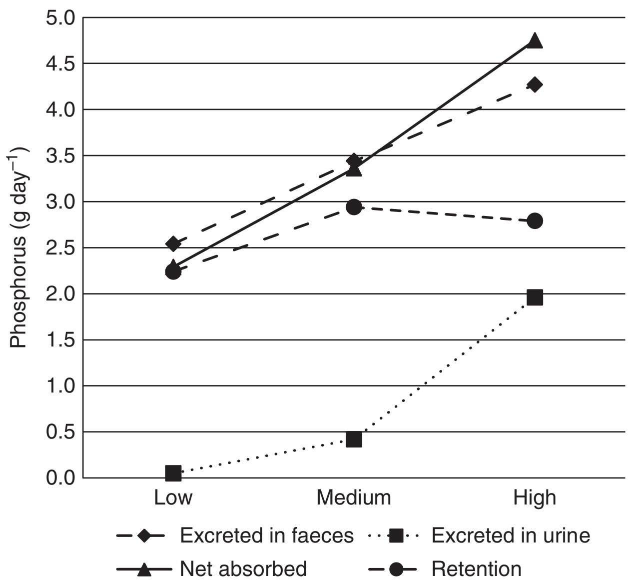

Fig. 3.10. The relationship between P intake, P absorption, P retention and P excretion in urine and faeces in the grower-finisher pig.

has also increased whereas the net P absorption still increases (Poulsen, 1994). It is obvious that a massive urinary P excretion indicates that the pig is fed excessive P amounts. A perfect P efficiency requires at the same time, a high P absorption (corresponding to a low faecal P excretion) and a low urinary P excretion. Figure 3.10 clearly shows that P absorption is not down-regulated to

a large extent when the pig is fed P in excess, leaving the urine as the main route for elimination of excess P.

The excretion and utilization of P in poultry depends on the type of the animal and production objectives. Table 3.4 shows that P utilization is the highest in the youngest broilers and decreases in heavier birds. Laying hens have a high P excretion and a low P utilization. Interestingly, birds raised organically have a higher P excretion compared with conventionally raised birds in Denmark. It is anticipated that the P efficiency will be greatly improved over the next few years due to increased focus on P use in poultry production.

Despite the wide use of microbial phytase supplementation in the feeding of monogastric farm animals, the utilization of P is often quite low and at the most about 60% in intensive farming. The main reason is that most studies on the improvements of P efficiency have been conducted using dry feeding systems in poultry and pigs. Studies with animals equipped with cannulas in the stomach have shown that the degradation of $InsP_6$ in dry fed pigs is not completely in the anterior part of the gastrointestinal tract before the place where the absorption of phosphate takes place. Approximately 60% and 30% of the phytate intake is still intact when the feed without or with microbial phytase leaves the stomach (Blaabjerg *et al.*, 2011). Such studies emphasize that the time needed for complete degradation of $InsP_6$ is more than 5 h. As such, the idea of 'pre-digestion' that takes place before the animal consumes the feed has been introduced. This approach is possible by use of liquid feeding where feed and liquids are mixed prior to feeding allowing sufficient time for the enzymatic degradation of $InsP_6$ to take place (Lyberg *et al.*, 2006; Blaabjerg *et al.*, 2010). Results have shown that complete degradation of $InsP_6$ before feeding may increase the P digestibility to about 70–75% depending on which feedstuffs are included in the feed (Blaabjerg *et al.*, 2010; Poulsen, 2012, unpublished results). However, the liquid feeding conditions such as time, pH, enzyme addition, temperature, mixing etc. have to be outlined to ensure safe use of this approach in practical farming.

Regardless of the worldwide use of phytase, a lot of inorganic P is still used as supplement in the feeding of monogastric animals. Obviously, the use of phytase is not appropriate in all cases, and it is anticipated that the use still has to be optimized in many farm situations. Furthermore, research on liquid feeding may create the basis for further replacement of inorganic P. This is really needed since the global P reserves are projected to be very limited and may cause a pronounced spike in P prices (Cordell *et al.*, 2009). In Denmark, the use of inorganic mineral P was calculated to approximately 13,000 t (elementary P) in 2001 regarding the feeding of monogastric animals (Poulsen and Rubæk, 2005). Since the introduction of a P tax in 2004, the use of feed phosphates has been reduced 40–50%. Another approach may be breeding for cereal or vegetable protein crops where phytate constitutes a reduced part of the total P content (Poulsen *et al.*, 2001).

Table 3.4. Phosphorus excretion and utilization in poultry (Poulsen *et al.*, 2012).

Type of poultry	Duration	Phosphorus excretion (kg)	Phosphorus utilization (% of intake)
1000 broilers	30 days	8.64	40
1000 broilers	32 days	9.71	40
1000 broilers	35 days	12.4	35
1000 broilers	40 days	16.2	34
1000 broilers	81 days, organic	41.1	15
100 female turkeys[a]	112 days	12.7	36
100 geese[a]	91 days	16.0	18
100 ducks[a]	52 days	4.29	32
100 layers hens, batteries	1 year	15.6	21
100 laying hens, free living	1 year	17.1	19
100 laying hens, organic	1 year	22.6	15

[a]The production of these birds are limited in Denmark and the calculated excretions are only indicative.

It is anticipated that the need for supplementation of feed phosphate will decrease over the next few decades along with the improvements in specific knowledge on the animals' P requirements (especially the reproducing animals) and upgraded experience and understanding of the use of microbial phytase. However, it is still forecast that globally there will be a need for feed phosphate supplementation in some cases, although the demand is anticipated to be greatly reduced. It is expected that very young animals and animals with a very low FCR may need extra P depending on the feedstuffs used. There are many different feed phosphates available on the market, but the P digestibility depends on the origin of the mined phosphate, the production principles, chemical composition etc. Many studies have found that the true P digestibility is around 80% in the water-soluble sources such as monosodium phosphate and phosphoric acid, about 54% in di-calcium phosphates and about 67% in mono-calcium phosphates (Poulsen, 2007). Higher values (up to 98%) are given in other studies but these values are expressed in apparent P digestibility (Dellaert *et al.*, 1990). The fact is that whenever feed phosphate supplementation is needed, feed phosphates with high P digestibility and low contents of undesired heavy metals such as cadmium, lead etc. should be preferred.

In conclusion, monogastric animals should be fed sufficient amounts of available P to cover the need to sustain maintenance and production to ensure health and growth/reproduction in a sustainable balance between welfare, production and environmental load. However, deficient but also excessive P supplies may have negative effects on bone mineralization and health (Sørensen *et al.*, 2012) as well as eggshell strength (Hammershøj, 2012, unpublished results).

Zinc and Copper Efficiency

Zinc (Zn) and copper (Cu) are essential trace elements, and at the same time they belong to the class of heavy metals. All monogastric animals require Zn and Cu, which normally are provided by the feed. Copper is required for the synthesis of haemoglobin and is part of many oxidative enzymes that ensure proper metabolic processes in the body. Zinc is part of many enzymes involved in the metabolism of carbohydrates, protein and fat. Furthermore, Zn also plays a role in cell replication and in the immune function. The physiological requirement for these minerals is not fully understood although most official tables on nutrient requirement specify values for most species and categories. However, these recommendations are given in total amounts per kg feed, because the knowledge on digestibility so far is too limited to be able to express the requirement in digestible amounts per kg feed. This was also the case for P until 20 years ago where the recommendations gradually shifted from total amounts of P to digestible amounts of P or amounts of non-phytate P in poultry (e.g. NRC, 1994, 2012; Pig Research Centre, 2012).

Copper has been widely used as growth promoter in pig production although the mode of action has not yet been fully understood. However, due to environmental concern, the use of Cu as growth promoter has attracted much attention and in some areas (e.g. within the EC), and the use of Cu as growth promoter has now been restricted and limited to be used in very young pigs after weaning (until 12 weeks of age). Zinc fed in excess amounts is known to induce moult in hens, but no reports on environmental concerns have been given so far. Zinc (as ZnO) has been widely included in high concentrations in feed for weaning pigs for about 25 years after a report showed positive effects of ZnO being able to prevent and cure post-weaning diarrhoea (Poulsen, 1989). It was shown that this high concentration of ZnO had to be given for 2 weeks until the pigs' feed intake was normalized. However, the use of high concentrations of ZnO was not restricted to 2 weeks post-weaning in many occasions. This has attracted environmental concern in livestock-dense areas due to high amounts of Zn accumulation in soils amended with manure from pigs fed Zn-fortified feed (Jondreville *et al.*, 2003). Different scenarios show that it may only take 50–200 years to reach a critical concentration of Zn and Cu in soils amended with pig manure in intensive pig-producing areas (Poulsen, 1998).

Methane Emissions

Over the last decades, the emission of greenhouse gases (GHG) has attracted much concern due to climate change concerns. Methane is produced during enteric fermentation and represents a

loss of energy to the animal. In slaughter pigs, the enteric methane production corresponds to 0.2–0.5% of the gross energy intake approximating to 3.4 l methane per day, but the loss will be higher in pigs fed more fibrous feed (Jørgensen *et al.*, 2011). The enteric production of methane in monogastric animals is very low compared with the losses in dairy cows covered in Chapter 2, this volume. However, the methane emission in the barn and during storage of pig manure may contribute markedly to the total emissions from the agricultural sector (Mikkelsen *et al.*, 2005). The calculated contributions from the pigs were 10% from enteric methane production and 70% from manure storage amounting to 26% of the total methane emission from the Danish livestock production (Mikkelsen *et al.*, 2005). Recently, experiments have shown that the emission of GHG could be lowered by acidification of the feed before feeding (Eriksen *et al.*, 2010) or by acidification of the manure (Petersen, 2012, unpublished results).

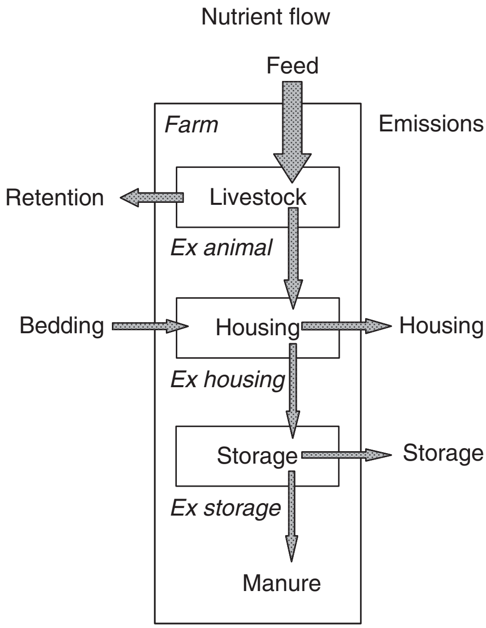

Fig. 3.11. The energy and nutrient flow on a farm (Poulsen *et al.*, 2006).

Summarizing Considerations

The nutrient flowchart (Fig. 3.11) illustrates that feeding affects all of the following steps from the feed to the final step where manure is amended to the soil. Consequently, the applied practical feeding condition and management enormously affect the production efficiency and utilization of energy and nutrients in the primary livestock production but also quantitatively on the losses of ammonia and GHG during housing and storage of manure.

Therefore, the feeding measures and the use of efficient pig and poultry breeds have huge impacts not only on the overall nutrient and energy efficiency but also on the quality of the manure used as fertilizer amendments to the soils used to produce cereal or protein crops. Table 3.5 summarizes the main factors (dietary means, feeding strategies, breeding and management) that affect the production efficiency and nutrient excretion and utilization.

Life cycle assessment (LCA) is often used to summarize the overall efficiency of productivity. Recently, such an approach was used to evaluate the environmental impact of the Danish pig production. The environmental assessment was performed using data representing the typical Danish pig production in 2010 on the one hand, and on the other using data from the 25% of Danish pig herds with the highest technical efficiency in terms of number of piglets per sow and feed use per kg pork produced (Nguyen *et al.*, 2011). The latter group of herds is expected to be representative for the typical Danish pig production in a few years. The LCA showed that by far the major part of the environmental impact occurs at the farming stage. This stage is responsible for more than 95% of the total impacts except for use of non-renewable energy, where the slaughtering stage and related transport accounts for 12–13%. The low impact related to the slaughtering stage is due to an efficient use of residues and by-products for energy production and feed (Nguyen *et al.*, 2011). Interestingly, the 25% of herds with the highest technical efficiency produce pork with a 10% lower environmental impact, indicating that this will be the case in a few years (Nguyen *et al.*, 2011).

Conclusions

In many ways, monogastric animals compete with humans for feed as they are primarily fed

Table 3.5. Main factors affecting the production efficiency and nutrient excretion in monogastric animals.

	Approach	Productivity	Nitrogen excretion	Phosphorus excretion
Dietary means	Protein balanced diets	Higher	High	
	Balanced dietary amino acid supply	↔	↓	
	Substitution of protein by industrial amino acids	↔	↓	
	Increasing digestibility of protein and amino acids	↔	↓	
	Increased phosphorus digestibility	↔		↓
	Replacement of feed phosphates by phytase	↔		↓
Feeding strategy	According to physiological requirement	↔	↓	↓
	Use of phase feeding	↔	↓	↓
	Optimization of feeding systems to avoid waste	↔	↓	↓
	Use of liquid feeding	↔	↔ (↓)	↓
Breeding	Selection programmes	↑ (kg gain per kg feed; litter size)	↓	↓
Management	Compilation of above mentioned methods	High productivity, welfare and product quality	↓	↓

on feedstuffs (cereals and vegetable protein sources) that may be used directly for human consumption. However, the demand for meat is predicted to increase in the future, which may put pressure on an increased use of by-products, not only in the feeding of ruminants but also the feeding of monogastric animals. Improvements in production efficiency have so far reduced the environmental footprints per kg of product in monogastric animals, and further improvements in production efficiency caused by genetic selection or by improvements in feeding management are expected to reduce the environmental impacts further. In future, these improvements in production efficiency will be balanced to comprise not only costs and environmental factors but also include health and welfare aspects in the primary production and socio-economic considerations in humans.

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4 Animal Welfare: An Integral Component of Sustainability

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Introduction

There has been a tremendous increase in the consumption of animal products over the past four decades (Steinfeld *et al*., 2006). It is predicted that this increase will continue, particularly as per capita income increases, with the greatest projected growth in developing countries (FAO, 2006; Steinfeld *et al*., 2006; Thornton, 2010). Concurrent with increased demand for food animal products, the developed world has experienced intensification of agricultural production over the last half century (Steinfeld *et al*., 2006). This intensification has resulted in polarized positions taken by both critics and defenders of the many issues associated with food-animal production systems (e.g. food availability, food quality, rural populations, use of resources, biodiversity and animal welfare; Fraser, 2008a; Croney and Anthony, 2011; Croney *et al*., 2012). Given these debates and the projected increase in the world's human population, combined with pressure for food costs to remain low, it is not surprising that an increasing number of questions are being raised concerning the sustainability of the world's food-animal agricultural resources.

Although there are many definitions of sustainability, it is generally accepted that environmental, social and economic considerations all play a role in this concept (Thompson, 1997, 2007). Environmental concerns are often what first come to mind when discussing sustainability, as evidenced by the considerable emphasis in this book (see Chapters 7, 9, 10, 11 and 12). Economic concerns, for example costs of producing and purchasing food-animal products, have been and will continue to play a key role in discussions on sustainability of animal agriculture (see Chapter 13, this volume). Social issues considered in the sustainability of animal agriculture are very broad, but include social justice, poverty and ethical consumerism (Croney and Anthony, 2011; Driessen, 2012). It is, however, important to note that social issues are not limited to the 'social' component in definitions of sustainability. Many of the concerns about the environment and economics of food production are rooted in public acceptability. In this way, public acceptability is of importance to all aspects of sustainable food production.

Animal welfare has emerged as one of the key public concerns regarding animal agriculture and clearly affects the sustainability of these production systems (Thompson *et al*., 2011). Indeed, the public perception and acceptability of farm animal welfare issues can, as we illustrate below, determine if specific husbandry practices and housing conditions are allowed to

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be used now or in the future. In this chapter, we explain what we mean by animal welfare and how both the social and natural sciences play key roles in understanding this complex field. Using egg production as an example, we will then illustrate how animal welfare concerns have played a driving role in policy and regulatory decisions regarding production practices, and arguably have played the key role in shaping the future of egg production. Given that the elements of sustainability overlap, we provide examples where synergy exists between improved animal welfare and the economic and environmental aspects of sustainability. In this section, we also discuss the challenges and complexities associated with the integration of animal welfare in sustainable animal agriculture.

What is Animal Welfare?

Concern about the care of farm animals is nothing new – farmers and veterinarians have always been concerned about the condition of animals in their care and have tried to ensure that they are healthy and well nourished. In this older tradition of animal care, good animal husbandry is seen largely as the provision of food, water, shelter and the absence of pain, illness or injury, and the focus is upon protecting the welfare of individual animals, ensuring that sick animals receive timely and effective care. However, beginning in the 1960s, the issue of whether farm animals were receiving *humane* care began to be questioned (Harrison, 1964). The timing of this increased scrutiny coincided with two primary changes in food animal production systems. The first was the growth of confinement systems, which resulted from technological advancements that were primarily justified from the perspective of economic efficiencies (e.g. reduced labour, refrigeration, land costs) and in some cases, such as egg-laying production, concerns about food safety. This in turn likely facilitated the second dramatic change, namely, significant increases in animal numbers on individual farms that ultimately resulted in reduced farm numbers (Fraser, 2008a). For example, over the span of three decades the swine industry in the USA, which during this period transitioned to nearly 100% confinement housing, lost over 88% of its producers but increased production by 10% (Vansickle, 2002). The intensification of animal production and subsequent public concern about animal care in these systems has resulted in broader concerns about animals in agriculture. It has become clear that animal welfare includes more than health alone, and other considerations, such as the extent to which animals are able to move and perform their normal behaviours, now play an important role in evaluation of the quality of animals' lives.

Modern-day concerns about animal welfare can be divided into three categories: (i) physical functioning, meaning that animals should be healthy and thriving, such that one function should not be enhanced to the detriment of another and behavioural and physiological systems should not be pushed to a point where health may break down; (ii) naturalness, meaning that animals have the ability to engage in behaviours that they are strongly motivated to perform; and (iii) subjective states, in that animals can enjoy life; that is to say, they experience positive states and that negative states (e.g. pain) are minimized (Fraser, 2008b). This can also be summarized as saying that concerns centre on care of their bodies, their natures and their minds (Appleby, 1999).

Intensification of animal agriculture has affected animal welfare in all three categories. In some cases, animal health has been negatively affected by intensification; selection for milk production in dairy cows and growth in broilers has resulted in a number of deleterious effects on health, including increased mastitis, lameness and reduced fertility for dairy cows and more musculoskeletal problems for chickens (Rauw *et al.*, 1998). In contrast, more intensive systems improve physical function by providing protection from extreme weather and better health management (e.g. vaccination programmes etc.). However, many intensive systems may prevent behaviours that animals are highly motivated to perform, such as nesting by laying hens (Cooper and Appleby, 2003) and sows (Tuyttens, 2005). Finally, many concerns about intensive animal systems centre on the subjective experience of the animals, for example, when they experience pain and suffering associated with standard procedures, such as castration or dehorning, because pain relief is not provided (Rault *et al.*, 2011; Stafford and Mellor, 2011).

Concerns about the negative aspects of intensification have led to the emergence of the field of animal welfare science. Researchers working in this field use a multi-disciplinary approach to understand all three of the broad categories of concern and have provided science-based information to address them (see books such as Fraser, 2008b; Appleby *et al.*, 2011). As an example, tail docking of dairy cattle (removal of the tail below the sixth or seventh vertebrae) emerged as a management practice in the US in the early 1990s (Johnson, 1992). Although adopted by many farmers due to perceived benefits, it was also criticized within and outside the dairy industry and thus became the subject of scientific study and has been the subject of scientific study within the last 20 years. Tail docking began for several reasons, including improved worker comfort and because it was thought to improve cow cleanliness and udder health by preventing transfer of debris and faeces from the tail to the udder. However, scientific evaluation of this practice has found that tail docking does not influence udder cleanliness (Eicher *et al.*, 2001; Tucker *et al.*, 2001; Schreiner and Ruegg, 2002), nor any aspect of udder health (Tucker *et al.*, 2001; Schreiner and Ruegg, 2002; Fulwider *et al.*, 2008). Although there is only limited evidence that the procedure causes pain, the ability to remove flies is impaired by tail docking (reviewed by Sutherland and Tucker, 2011). The importance of the scientific information gained from this type of research is evident from the extent to which it has now been incorporated into industry codes of practice, retailer standards and legislation (Appleby, 2013; Mench 2008). As a result of the scientific evidence and public concerns about tail docking (Weary *et al.*, 2011), for example, this practice has been banned or discouraged in most countries (Sutherland and Tucker, 2011). Science will hopefully continue to inform decisions about all aspects of animal production from entire housing system (see section 2) to specific management practices.

Animal Welfare as a Driver of Sustainability: the Egg Industry

The important role that animal welfare concerns can play in shaping how animals are produced, and thus the sustainability of production systems, is illustrated by the recent changes in the egg industry. Worldwide, the vast majority of commercial egg-laying hens are housed in conventional (battery) cages (Mench *et al.*, 2011). The conventional cage system was developed in the 1930s as an alternative to free-range and barn production systems, which exposed hens to health and predation risks. Conventional cages were also more efficient economically than barn and free-range systems, such that in a span of 30 years the majority of hens in developed countries were housed in these cages. However, as the use of conventional cages grew, so too did public concerns about the welfare of the hens, mainly because of the extent to which these small, barren enclosures restricted the behaviour of the hens. In effect, concerns about hens' physical functioning (e.g. health) had been dealt with by placing them in a housing system that restricted their ability to perform their natural behaviours to such an extent that the public in some countries found that system to be unacceptable.

As mentioned above, animal welfare began to receive more attention in the 1960s, as the result of the publication of Ruth Harrison's book *Animal Machines* (1964) and the subsequent release of a UK government report (Brambell, 1965) on intensive animal farming systems. The so-called Brambell report was extremely critical of production systems that restricted the movement of animals such that that they could not sit, stand, lie down, turn around and/or groom without restriction. By 1976, the Council of Europe had developed the *Convention on the Protection of Animals Kept for Farming Purposes*, which stated that tethered or confined farm animals should be given space appropriate to their physiological and ethological (behavioural) needs (Appleby, 2003). In keeping with the language of the Convention, the European Union adopted minimum space standards for hens kept in conventional cages. However, despite this public concern continued to grow (Savory, 2004) and in 1999 the European Union banned conventional cages, effective 2012. Subsequently, several European countries independently passed legislation that either banned the use of these cages immediately or required additional cage or stocking density modifications (Appleby, 2003; Jendral, 2005).

Similar issues have now arisen in the USA, where there is currently no framework for federal

regulation of the food animal industries with respect to animal welfare. Despite the widespread adoption in 2000 by the trade organization of the egg industry, the United Egg Producers (UEP), of animal welfare standards specifying (among other things) minimum space allowances in conventional cages (UEP, 2010), pressure continued to mount on the egg industry to move to alternative production systems (Mench *et al.*, 2011). This pressure came mainly in the form of voter initiatives and regulations at the state level, promoted in large part by the Humane Society of the United States (HSUS).

Because of the potential for economic disruption of the industry as a whole due to inconsistent and conflicting state regulation, the UEP and the HSUS decided in 2011 jointly to seek federal regulation focusing on laying hen welfare (HSUS, 2011). If passed, this regulation would codify various management practices that are already part of the UEP standards, but in addition outlaw conventional cages. The alternative systems that will be allowed include non-cage systems (e.g. aviaries, free-range) and enriched colonies. The latter system was developed in Europe in the years following the initial announcement of the conventional cage ban in response to concerns about some of the persistent negative hen health aspects of non-cage systems (Lay *et al.*, 2011), and with considerable input from animal welfare scientists (Mench *et al.*, 2011). Enriched colonies (also called furnished cages) are larger than conventional cages in order to provide sufficient space to allow hens more freedom of movement, and contain perches, a nesting area and a foraging area to satisfy hens' needs to perform pecking and scratching behaviours.

It seems likely that the use of conventional cages will come under increasing scrutiny and regulation in developed countries. For example, the National Animal Welfare Advisory Committee of New Zealand, a committee established under the Animal Welfare Act to provide advice to the Ministry of Agriculture, recommended that conventional cages be phased out, which the Ministry has agreed producers will be required to do by 2012 (www.biosecurity.govts.nz). The egg farmers in Manitoba, one of the Canadian provinces, have decided to discontinue using conventional cages beginning in 2018 due to public concerns, to be replaced by furnished cages (Friesen, 2010). Together, these examples of legislation and industry-led initiatives illustrate how public concern about animal welfare can be the primary driving factor in the sustainability (or lack thereof) of animal production systems.

Challenges and Complexity: Animal Welfare and Other Aspects of Sustainability

Given that both animal welfare and sustainability are multi-faceted concepts, it is unsurprising that the relationship between the two ideas is complex. There are examples where improvements to animal welfare can support or, alternatively, conflict with other aspects of sustainability. The following section outlines examples of these situations with respect to economic and environmental issues. The examples are limited to evidence-based concerns about animal welfare and focus on animal interactions with the environment or management practices. Other aspects of changing housing systems or management practices associated with animal welfare, such as worker satisfaction and health and food safety are also important, but are beyond the scope of this chapter. This section will end with a discussion of how animal welfare can be incorporated into different sustainability criteria and evaluation.

Economic sustainability and animal welfare

There are numerous management practices that improve both animal welfare and economic sustainability. For example, appropriate animal handling by humans is an important component of good welfare and plays a role in the productivity, and thus economic sustainability, of farming systems. Negative handling practices, such as hitting and shouting, are associated with lower growth rates and milk production in pigs and dairy cattle, respectively (reviewed by Hemsworth, 2003). Indeed, rough handling can decrease meat quality at slaughter for both pigs and cattle. For example, use of electric prods increases the prevalence of pale, soft and exudative (PSE) pork meat, as a result of increased post-mortem acidification

(D'Souza *et al.*, 1998). Handler behaviour is also associated with udder health, such that cows that have more positive interactions with milkers have fewer infected quarters (Ivemeyer *et al.*, 2011). Cattle will clearly show avoidance of negative handling if given the choice, indicating they find it aversive (Pajor *et al.*, 2003). Improved handling is thus beneficial for both animal welfare and economic performance of swine and dairy farms, and, in this way, several aspects of sustainability converge.

There are other cases where management practices seem likely to improve animal welfare and economic sustainability, but establishing the explicit link in terms of economic analysis has only recently begun to receive attention and thus there is only limited work available. For example, dairy replacement heifers are often fed a limited amount of milk (e.g. 10% of body weight) during the first 5–8 weeks of life in order to encourage intake of solid food. However, dairy calves provided free access to milk will typically consume more than twice this amount (reviewed by Khan *et al.*, 2011). Calves fed more milk grow faster (~1 versus ~0.45 kg day^{-1}), exhibit fewer signs of hunger, show less abnormal behaviour and are less likely to compete for food in group situations. Moreover, young dairy calves are physiologically unable to digest solid feed before 2–3 weeks of age and thus calves fed more milk have better feed efficiency and less disease, all changes indicative of improved welfare. In addition, there are thought to be longer-term economic benefits of feeding calves more milk. Higher growth rates are associated with earlier breeding age, thus reducing the amount of time until heifers produce milk, and higher milk yield after parturition (e.g. Soberon *et al.*, 2012). Thus, feeding calves more milk improves welfare and may be beneficial from an economic perspective given the financial gain associated with the longer-term benefits described above.

Last, there are cases where making changes to address welfare concerns markedly increase the cost of production, and thus have potentially negative effects on economic sustainability. This is most often seen when there are major changes in housing systems. The move from conventional cages to alternative systems for laying hens discussed above is one such example. Replacing conventional cages not only requires significant capital investment, but operating costs for non-cage systems are also higher, by as much as 40–70% (Sumner *et al.*, 2011), mainly due to the increased feed consumption of hens in these systems (Xin *et al.*, 2011).

Producers will, of course, be buffered from negative economic impacts if consumers are willing to pay the premium for the additional costs of alternatively produced eggs (or other animal products). However, consumer behaviour is very complex, and consumers' purchasing patterns often fail to correspond to their stated purchasing preferences. Surveys in Europe and the USA find that most consumers of animal products are concerned about animal welfare and are willing to pay more to purchase products that they perceive to be more 'welfare friendly' (Norwood and Lusk, 2011; Thompson *et al.*, 2011). However, this may not translate to consumer behaviour in the marketplace, where economic considerations often take precedence over concerns about animal welfare or other aspects of sustainability (Bennett and Thompson, 2011). The reason for these disconnects between consumers' stated values and their purchasing behaviour is complex and still not well understood. For some animal products, it may be that consumers do not actually have sufficient information to make informed purchasing decisions (e.g. adequate labelling), that they do not have an acceptable range of choice in their local markets, or that the retail premium for 'alternatively' produced products is so much greater than the actual cost of producing those products that less affluent consumers are deterred from buying them (Norwood and Lusk, 2011). It may also be that many consumers feel that providing social goods, for example improving animal welfare or protecting the environment, should be accomplished via extra-market mechanisms (e.g. regulation) rather than market mechanisms (Schröder and McEachern, 2004; Norwood and Lusk, 2011).

Environmental sustainability and animal welfare

Environmental issues receive considerable attention in discussion of sustainability, and for some animal production systems these have been the primary public concern. Improved animal welfare and environmental sustainability can converge, but discussion of these two issues together

highlights the complexity of farming systems. Providing pigs with straw is an example where animal welfare and improvements in environmental sustainability can go hand in hand. Straw has been shown to be advantageous for growing pigs in a number of ways (Tuyttens, 2005); its use improves thermal comfort while lying and provides opportunities for exploration, rooting, foraging and chewing. Providing alternatives for pigs to engage in oral activities are important to prevent injurious behaviours such as tail biting (reviewed by Schrøder-Petersen and Simonsen, 2001; Taylor *et al.*, 2010). However, there are practical constraints associated with incorporating straw into intensive systems, as the manure handling system must be compatible. One alternative housing system, the Straw-Flow system, contains two distinct areas, one area for lying and oral activities that contains straw and another, strawless area for excreta. This system capitalizes on the willingness of pigs to use a dunging area that can be frequently cleaned. Pigs kept in the Straw-Flow system show behavioural changes consistent with improved animal welfare (e.g. less pig-directed oral behaviour; Kelly *et al.*, 2000). Greenhouse gas emissions (CH_4, N_2O, NH_3) are also lower in Straw-Flow systems, compared with reference values for forced ventilated, fully slatted (thus strawless) floor systems (Amon *et al.*, 2007). Thus, in this type of system, animal welfare and environmental sustainability are both improved by the addition of straw in this housing design. Moreover, Bornett *et al.* (2003) argued that that adding straw only provides a modest increase in cost (7%), indicating that economic sustainability (at least in the UK) may also correspond in this situation.

This relationship is not always so straightforward. As discussed above, one specific animal welfare concern expressed by the public is animals that are kept under 'unnatural' conditions with limited space and often a limited ability to engage in social interactions and other natural behaviours (Fraser *et al.*, 1997). The problem with environments where animals are prevented from being able to stretch their wings or limbs or turn around freely can only be addressed by less restrictive housing, such as group housing systems for gestating sows or more space for laying hens. Sow housing provides an example of how complex comparisons between intensive and less restrictive systems can be from a welfare and environmental perspective.

Sows are often housed in gestation stalls for most of the duration of their 4-month pregnancies and in farrowing crates, where they give birth to their young (Fraser *et al.*, 2001). By design, these types of housing reduce the challenges associated with aggression between sows during gestation and crushing or savaging of newborn piglets at farrowing, both important welfare concerns. However, these forms of restrictive housing place limitations on freedom of movement. It is estimated that 92% of sows in these systems of confinement exhibit oral stereotypies, which are abnormal behaviours associated with confinement (Mason and Rushen, 2006). In contrast, outdoor housing allows sows a high degree of freedom of movement (Hötzel *et al.*, 2004) and the ability to express a larger repertoire of natural behaviours that they are highly motivated to perform, such as rooting and nesting behaviour (e.g. Špinka, 2006). Outdoor housing systems for sows were previously criticized for having higher rates of neonatal mortality (Ngapo *et al.*, 2004), but current estimates of live-born mortality in the UK are similar for outdoor-housed sows (10.5%) and indoor-housed sows (11.8%, Meat and Livestock Commission, 2006). Despite the welfare benefits of outdoor systems, there are a number of environmental implications of this type of housing (reviewed by Siegford *et al.*, 2008). An obvious limitation is weather and exposure to fluctuations in environmental temperature (Akos and Bilkei, 2004). Outdoor systems are only viable in regions that do not have extreme weather and, even then, may not be as efficient as indoor housing, in terms of environmental impact. For example, nitrogen and phosphorus surpluses are highest on Danish farms that keep sows outside (Nielsen and Kristensen, 2005), although management and stocking density likely affect the degree of this problem (Williams *et al.*, 2000). More intensive housing systems also face challenges with nitrogen containment (e.g. Karr *et al.*, 2001), but non-point sources of contamination may be more difficult to control in outdoor systems than in more intensive ones. There are also concerns about land use associated with outdoor housing systems for sows. Namely, more space is needed for these systems and allowing pigs to root reduces pasture cover and can intensify

nutrient leaching (discussed by Hermansen *et al.*, 2004). Concerns about pasture damage can be addressed by using nose rings, which prevent rooting by making it painful to press the snout against the ground (Horrell *et al.*, 2001). Thus, even within outdoor systems, trade-offs may be made between the welfare of the animals (e.g. their ability to engage in rooting behaviour use of painful nose rings) and environmental concerns (e.g. pasture quality, damage, nutrient leaching). This example highlights the need to consider all aspects of a production system, including both welfare and environmental implications, in order to assess sustainability.

Animal welfare and assessment of sustainability

Assessments of sustainability and animal welfare are riddled with difficulty, both when considered separate issues (e.g. environmental sustainability or animal welfare alone) and in combination with each other. This difficulty arises because decisions about what is sustainable depend not just upon science, but also upon societal values. With environmental issues, for example, let us say that a (hypothetical) alternative housing system for animals is found to have positive effects on air quality but negative effects on water quality compared with conventional systems (as might be true for free-range compared with confined poultry production). The question then arises as to how to weigh the relative importance of these two aspects of environmental impact, a question that can only be resolved by considering values. Similar kinds of problems arise when evaluating changes that have multiple economic impacts, for example economically advantaging some producers (say, large producers) while disadvantaging others (say, small producers).

With respect to animal welfare, similar types of conflicts in assessment arise because individuals vary in the extent to which they value the various aspects of welfare, such as physical functioning or naturalness. Again, conventional cages for hens are an example of this kind of conflict. Hens in conventional cages have an overall better health status than hens in non-cage systems, but are severely restricted behaviourally (Lay *et al.*, 2011). If physical functioning is weighted more heavily than naturalness in assessments of welfare then conventional cages are the preferred system, but if naturalness is weighted more than physical functioning then alternative systems like cage-free or free-range will be preferred. These differences in values are often borne out in public policy. For example, as of 2012 the European Union has banned conventional cages but still allows enriched cages; however, specific European countries, such as Austria, Belgium and Switzerland, have either already banned both conventional and enriched cages or plan to do so (Kerswell, 2011). Both the European Union and the governments of these countries had access to the same scientific evidence evaluating hen-housing systems but weighed public values differently, thus leading to different public policy decisions. This, of course, also raises questions about the sustainability of the enriched colony housing system being proposed for adoption in the USA, given that it is still unclear whether this system will satisfy public values about hen welfare.

As with sustainability in general, there has been increasing interest in developing scientifically based metrics and weighting systems to assess animal welfare. This has been driven in large part by the growth of certification programmes that require farms to be scored in order to determine if they pass or fail animal welfare audits (Mench, 2008; Swanson *et al.*, 2011). These approaches, which often include identifying relevant welfare measures based on experimental and on-farm scientific data and then ranking and integrating those measures based on expert opinion, are useful steps forward but cannot eliminate value judgements (Swanson *et al.*, 2011). Because of the value judgements inherent to all 'wicked' problems like assessment of animal welfare or sustainability, it has been suggested that participatory integration strategies, which involve dialogue among stakeholders, may be the most promising approaches to building consensus about the relative importance of sustainability criteria (Swanson *et al.*, 2011; Thompson *et al.*, 2011).

Participatory integration strategies have been used to incorporate animal welfare considerations into sustainability assessment. Mollenhorst and de Boer (2004) used a participatory analysis involving a heterogeneous group of stakeholders, including retailers, egg

producers, non-governmental organizations and policy makers, to identify issues important to egg production in the Netherlands. The major sustainability weaknesses, strengths, threats and opportunities identified during the brainstorming session for egg production in conventional cages related to animal welfare and health, environment, egg quality, ergonomics, economics, consumer concerns, and knowledge and innovation. Sustainability aspects of different egg production systems were quantified with reference to issues identified both by this type of participatory analyses and literature reviews (de Boer and Cornelissen, 2002). This process involved selecting indicators that were measurable and then assigning a target value for each based on political goals, scientific knowledge (Mollenhorst *et al*., 2006), or expert judgement. Once such values are assigned, the contribution of each individual indicator to the overall sustainability is determined mathematically. De Boer and Cornelissen (2002) used just such a method to compare different hen-housing systems, by assigning values for measurable aspects of important economic, ecological and social (animal welfare, egg quality, farmer welfare, and odour nuisance) concerns and then creating a sustainability index. Although such approaches are useful attempts at integrating information to arrive at decisions about sustainability, at some point they also involve making value judgements about relevant indicators, appropriate measurement methods and/or how to rank or weight different indicators (Swanson *et al*., 2011).

Indeed, some concerns about aspects of modern animal agriculture are not easily addressed by either science or assessment schemes. The industrial scale of animal agriculture, for example, may always face challenges in terms of public acceptability. For example, large-scale farms offer economic advantages, but limit the amount of attention an individual animal receives. A typical cage layer house in the USA can contain 125,000 or more hens. If workers were to look at each bird for only 1 s day^{-1}, this evaluation would take 35 h day^{-1}, considerably more worker-hours than are currently devoted to this aspect of management. Although it would be possible to increase the amount of attention each hen receives by hiring more workers, one of the major economic efficiencies associated with large-scale egg production comes about because of the reductions in labour cost resulting from increased automation. In addition, this is a conservative estimate of the labour and possible financial cost required for individual care of each hen because even catching and euthanizing sick hens identified in the process would clearly take longer than 1 s per animal. For some non-poultry species large farms may be able to use technology to monitor individual animals to improve disease detection. However, this solution is dependent on the production system. Finally, large farms may well be situated to provide benefits such as consistent feed quality, skilled labour, and to undergo third-party auditing schemes. The issues inherent to large-scale animal agriculture (e.g. individual care, concentration of manure in a single location) are important considerations in any further intensification in the name of sustainability. A long-term, comprehensive approach that takes into consideration public acceptability is needed in any discussion of sustainable animal production systems and, to date, there is no assessment scheme that encompasses all of this complexity.

Conclusions

Concerns about environmental, economic and social impacts of food production are all rooted in public acceptability. Animal welfare is emerging as one of the key public concerns regarding animal agriculture. In this way, animal welfare is an integral component of sustainability. Public concerns about the humaneness of food animal production can result in banning of specific practices or even elimination of entire housing systems. The scientific study of animal welfare informs public policy and regulation about animal welfare, but is often given differing levels of emphasis based on the values and priorities of various stakeholders.

The challenges associated with considering all aspects of sustainability, including animal welfare, have only begun to be addressed. For example, when considering the relationship of animal welfare to other aspects of sustainability, there are examples where improvements to animal welfare can support or conflict with regards to these other types of concerns. To address this complexity, sustainability assessment schemes have begun to incorporate animal welfare with other aspects of sustainability,

and stakeholder input has been used to weight the importance of each element. However, attempts to 'achieve' sustainability may be viewed differently by various stakeholders. This challenge, the difference between how the various components are valued, highlights the importance of viewing animal welfare and sustainability as a continuum. In this view, the goal is continued improvement on all aspects of the assessment. This approach will help identify ways to improve animal welfare and sustainability of existing animal production systems.

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5 Genetics and Sustainable Animal Agriculture

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Introduction

Genetics may not be immediately associated with the term sustainability, and yet the importance of animal genetics in contributing to the interplay between the environmental, social and economic goals of sustainability should not be underrated. Genetic gains are both permanent and cumulative meaning that gains made in 1 year will be transmitted to subsequent generations without further endeavour or expenditure. Genetic improvement has been an important component of the tremendous advances in agricultural productivity that have occurred over the past 50 years. Perhaps this is nowhere more evident than in poultry breeding (Table 5.1). The body weight of broiler (meat) chickens at 8 weeks of age has increased from 0.81 to 3.14 kg between 1957 and 2001, and approximately 80% of this fourfold increase was due to genetic selection (Havenstein *et al.*, 2003). Increased productivity clearly benefits the economics of production. Animals that can be grown to market weight at a younger age use proportionally less of their total feed intake on maintenance energy. In 1960, the average time needed to produce a broiler chicken in the USA was 72 days. By 1995, this was reduced to 48 days, including an increase in average slaughter weight of 0.4 kg as is dramatically illustrated in Fig. 5.1. Concurrently, the feed conversion ratio (kg feed per kg gain) was reduced by 15%. These remarkable improvements in production efficiency have resulted in dramatic reductions of the inputs required to produce a kilogram of chicken. However, it has been argued that this was achieved without adequately considering important social and animal welfare components of sustainability.

From an environmental perspective, genetic improvement over the past 50 years has also resulted in reductions in greenhouse gas (GHG) emissions and global warming potential per unit of animal product (Table 5.2). Capper *et al.* (2009) reported that although the carbon footprint per individual cow increased when comparing 1944 with 2007, due to increases in the milk production per cow, the carbon footprint per unit of milk in 2007 was 63% lower than in 1944. It has been observed that despite intense selection on specific traits (e.g. 8-week body weight in broiler, milk yield in dairy cattle) the selection response per generation for these traits shows no sign of decreasing. Mean milk yield in the USA has more than doubled during the 50 years starting in 1957 from 5,859 to 12,043 kg year^{-1} per cow, with a rate of 1% yield increase per generation. A similar trend

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Table 5.1. Typical broiler performance in the USA (Gordon, 1974[a]; Havenstein *et al.*, 2003[b]).

Year	Weeks of age when sold	Live weight (kg)	Feed efficiency (kg feed per kg gain)	Mortality (%)
1923[a]	16.0	1.00	4.7	18.0
1933[a]	14.0	1.23	4.4	14.0
1943[a]	12.0	1.36	4.0	10.0
1953[a]	10.5	1.45	3.0	7.3
1963[a]	9.5	1.59	2.4	5.7
1973[a]	8.5	1.77	2.0	2.7
1957[b]	12.0	1.43	3.84	4.7
2000[b]	6.0	2.67	1.63	3.6

Fig. 5.1. Contemporary comparison of (a) 1957 control and (b) 2001 selected broiler carcasses slaughtered at different ages (from left; 43, 57, 71 and 85 days.). Photo by G.A. Havenstein. Permission for use of Fig. 5.1 granted from the Annual Review of Ecology Evolution and Systematics, Volume 41, 1–19. ©2010 by Annual Reviews www.annualreviews.org.

Table 5.2. Proportional changes (%) in greenhouse gas (GHG) emissions and global warming potential (GWP_{100}) per unit of animal product achieved as a result of 20 years (1988–2007) of genetic improvement as calculated by Department for Environment, Food and Rural Affairs (Jones *et al.*, 2008).

	CH_4	NH_3	N_2O	GWP_{100}
Chickens – layers	–30	–36	–29	–25
Chickens – broilers	–20	10	–23	–23
Pigs	–17	–18	–14	–15
Cattle – dairy	–25	–17	–30	–16
Cattle – beef	0	0	0	0
Sheep	–1	0	0	–1

has been observed in a long-term maize selection experiment, where oil content has consistently increased 3% per generation for 100 generations (Dudley and Lambert, 2004). It is likely that this sustained response is fuelled by new mutations that arise each generation (Hill and Kirkpatrick, 2010).

Since the early 1960s, livestock production has grown rapidly with a worldwide fourfold increase in the number of chickens, twofold

increase in the number of pigs, and 40–50% increases in the numbers of cattle, sheep and goats. The global number of livestock animals used in agricultural production has been estimated to be 1.8 billion large ruminants, 2.4 million small ruminants (sheep and goats), 20 billion poultry and nearly 1 billion pigs (Niemann *et al.*, 2011). This so-called 'livestock revolution' is being driven by the sharp rise in demand for animal food products in many developing countries, resulting in a pronounced reorientation of agricultural production systems (Delgado, 2003). The United Nations Food and Agriculture Organization (FAO) predicts the global population will rise to approximately 8 billion people by 2030, and will exceed 9 billion people by 2050 (FAO, 2009). Accordingly, the demand for animal protein is also expected to grow as consumers in developing countries become more affluent. When the year 2000 is used as a base, projections indicate the need for a 68% and 57% increase in global meat and dairy consumption by 2030, respectively (Steinfeld and Gerber, 2010).

Although some may consider that the challenge of feeding the burgeoning world population would best be met by reducing livestock production and avoiding meat consumption, it is unlikely that vegan diets will be acceptable for many people. Although there are modest decreases in meat consumption in some developed countries, there is a widespread and growing preference in developing nations for dietary animal protein. Animal products contain concentrated sources of high quality protein that complement those of cereal and other vegetable protein. They also contribute dietary sources of calcium, iron, zinc, and several B group vitamins. There is evidence that increasing foods of animal origin in the diet of young children with an initially low rate of consumption of these foods leads to marked improvements in both physical and mental development (Neumann *et al.*, 2007). Additionally, some livestock, particularly ruminants, eat feedstuffs and graze marginal lands that are not appropriate for the production of plants that can be consumed by humans. Livestock can also provide a variety of goods and services that generate income and support the livelihoods of millions of poor people in the developing world. Their uses include livestock-derived food and products, sale of these products for income, use as assets for savings or trade, draught power and transportation, manure for soil fertility restoration, sale of livestock products for income, as a source of draught power and transportation, in diversifying livelihood options to reduce vulnerability, and their contribution to the socio-cultural roles and obligations of their owners (Rege *et al.*, 2011).

What is Animal Breeding?

Animal breeders use information (e.g. production data, pedigree records) and technology (e.g. DNA information) to select which animals will become parents of the next generation. The breeding value (BV) of an animal is defined as the superiority of its offspring when compared with the population mean. This value is estimated based on the pedigree and performance records of an animal and its relatives using mixed-model equations first described by Henderson (1953). Selection is based on the estimated breeding value (EBV) of the animal as it relates to the traits included in the breeding objective (BO). The BO can be thought of as the overall goal of the breeding programme. The role of the animal breeder is to maximize the selection response towards the BO. Response is defined as the difference in the mean phenotypic value between the offspring of the selected parents compared with that of the whole of the parental generation before selection. The genetic gain ($\Delta G\ t^{-1}$) per unit of time in animal breeding programmes is directly proportional to the accuracy of selection (correlation between an animal's EBV and true BV), selection intensity (the proportion of animals that are used as the parents of the next generation), the genetic variability (how much additive genetic variance there is in the population), and is inversely proportional to the generation interval (the time it takes to replace a generation). Any technology that can increase the selection response per unit of time will accelerate genetic progress towards the BO.

Multiple-trait selection indexes can be developed to optimize profit given a specific BO, with different traits being assigned an economic weight based upon their contribution to profit. The index ranking of an animal is equivalent to the term 'fitness' in wild populations, with the

highest ranked individual being the 'fittest' or most profitable according to the BO and therefore a desirable parent for a given production system. With classical index selection, the BO determines the targeted direction of genetic change for the traits, weighted by their respective market values (MV). This MV is the economic value per unit increment in the trait (e.g. \$ kg^{-1}, \$ per egg). The breeding goal (H) or aggregate genotype can be represented in the following equation:

$$H=MV_1EBV_1+MV_2EBV_2+\ldots+MV_nEBV_n$$

where EBV_i is the additive genetic value of trait *i*, and MV_i is the MV (also known as economic value) of trait *i*, defined by the change in profit of a unit change in the trait *i* (Hazel, 1943). The values for EBV_i are derived using a multi-trait best linear unbiased prediction (BLUP) computer program that takes into account genetic and environmental correlations among traits as well as pedigree relationships (Lynch and Walsh, 1998). Clearly animals that have the highest EBV for traits with a high MV will have the highest index value (i.e. will be the 'fittest' in this selection scenario) and will be most likely to become parents of the next generation.

Genetic improvements in efficiency have been most pronounced in those industries that have a highly structured breeding sector (e.g. dairy, pig and poultry) and well defined, profit-maximizing BO. These gains have largely come about because improved efficiency is directly associated with improved profitability. A small number of animal breeding companies control the genetics of these vertically integrated industries. For example, over 90% of global poultry breeding stock is managed by three companies selling to a worldwide market (Flint and Woolliams, 2008). Industries that have less vertical integration (e.g. beef and sheep) have made less progress. Animal breeding in these industries tends to be driven by breed associations, and because the MV of traits differs among industry sectors (e.g. breeder, farmer, feeder, processor), it is difficult to develop a single, industry-wide BO that is economically rational for all sectors. This leads to an important concept in animal breeding, the role of the 'decision maker' in animal breeding (Olesen *et al.*, 2000). In the absence of vertical integration, breeding goals will be developed based on the individual producer's financial interests. The producer is the one investing in breeding stock and in a competitive market their decision will be based on the ways they perceive that animals contribute to farm profit. If there is market failure in terms of attributes of sustainability (e.g. there is no price incentive associated with the inclusion of improved animal welfare in BO), then alternative approaches will need to be implemented to incentivize the inclusion of such considerations into BO in less vertically integrated industries. These incentives may take the form of subsidized breeding, regulations, fines for poor welfare or increased prices for products labelled according to specific welfare grades (Nielsen *et al.*, 2011).

Sustainable Animal Breeding

Key words characterizing sustainable animal breeding are *product quality, genetic diversity, efficiency, environment* and *animal health and welfare* (Nielsen *et al.*, 2006). The traditional methods used to derive MV in the BO with the objective of maximizing the profit of the farmer have focused on production traits such as milk yield, growth rate and meat yield. Key social goals such as food safety, food quality, environmental protection and animal welfare have often not been overtly included in BO. Torp-Donner and Juga (1997) reviewed studies that used different criteria to describe sustainable livestock production from the perspective of animal breeding. Their review suggested that under low and intermediate production levels, increased yield and efficiency will be more environmentally sustainable than goals associated with extensive production. However, intensive systems are associated with their own environmental concerns. Other stakeholders would like to see more selection emphasis placed on secondary or functional traits that are not directly associated with production outputs or even economic return, including traits that could lead to improved animal welfare and reduced GHG emissions. Several authors have discussed approaches to incorporate 'sustainability traits' into BO. As might be predicted from the rather broad definitions of sustainable animal breeding above, approaches vary depending upon which components of sustainability are under discussion.

One approach is to include functional traits that fall under the general heading of 'sustainability' in BO, using methodology developed for deriving profit-maximizing selection indexes. Traits that might fall into this category include disease resistance, reproduction, longevity and well-being. These functional traits are related to animal health and welfare and should arguably have been included in the BO from the beginning as they ultimately affect the profitability of an enterprise. Where these traits have declined as a result of selection it is generally due to their absence in the BO. There are several reasons why they have not been included in the BO up to this point; low heritability (the proportion of observed variation that can be attributed to inherited genetic factors in contrast to environmental ones), lower MV than production traits and/or negative genetic correlations with production traits. A negative genetic correlation between two traits means that selection for one trait (e.g. production traits) results in an unwanted change in the second trait (e.g. functional traits). Some examples include single trait selection for milk yield resulting in reduced resistance to mastitis and fertility in dairy cows, selection for increased growth rate resulting in increased leg problems and ascites (pulmonary hypertension) in broiler chickens, and selection for lean meat production resulting in reduced stress tolerance in pigs (Rauw *et al.*, 1998).

Including functional traits or other traits in BO will reduce the selection pressure (and hence slow genetic progress) in production traits. Production traits typically possess both high heritability and high MV, and so low heritability, low MV functional traits need to be given an inflated emphasis in the selection index to achieve the same rate of genetic progress, or at least to minimize their decline. Animal breeders are always negotiating trade-offs among competing goals, especially when it comes to breeding for sustainability (Gamborg and Sandøe, 2005). Typically it has been difficult to obtain records on functional traits, and the old adage that 'you can't manage what you do not measure' is particularly true for animal breeders. This raises an important point as it relates to breeding for sustainability. Animal breeding programmes that involve complex traits such as robustness, animal well-being, or disease resistance in the selection objectives require well defined phenotypes (records) upon which to base selection decisions. Identifying a phenotype that can be observed with high repeatability (test–retest reliability) and which can be used as selection criteria to quantify complex functional traits in the BO can be difficult, and ideal traits may be very expensive or impractical to measure. In that regard, an important advance will be development of objective, quantifiable measures of welfare, which could be used as selection criteria for breeding decisions (Hume *et al.*, 2011). Functional genomics has the potential to provide biomarkers, which can be used to define complex traits such as behaviour, stress, or disease in unique quantifiable ways (Kadarmideen *et al.*, 2006).

Some important examples where functional traits have been added to BO include the incorporation of fertility and disease resistance traits into dairy cattle selection indexes, and the inclusion of leg traits into poultry breeding. Table 5.3 shows the ten dairy traits that are currently included in the US dairy selection index

Table 5.3. Year that genetic rankings began and emphasis placed on dairy traits in 2010 US national dairy selection indexes.

Trait	Year begun	Emphasis (%)
1. Milk	1935	0
2. Milk fat	1935	16
3. Milk protein	1977	19
4. Calving ease/stillbirth	1978/2006	5
5. Udder shape and support	1983	7
6. Feet and leg conformation	1983	4
7. Body size/weight	1983	–6
8. Productive life/longevity	1994	22
9. Mastitis resistance	1994	10
10. Daughter pregnancy rate/fertility	2003	11

with the year that each trait was introduced. It can be seen that production traits (milk and fat) were the first traits incorporated into selection programmes. As time went on, more functional traits were included to widen the scope and redirect the emphasis of breeding programmes. At the current time, production traits represent 35% emphasis within the BO, with the remaining 65% placed on functional traits (Cole *et al.*, 2009). The genetic correlation between body weight and incidence of leg disorders in broilers is positive, so appropriate multi-trait selection indexes have been developed to permit a genetic improvement in leg health concurrently with continued, though more modest, improvement in growth rate. In recent years, genetic selection has had a major impact on decreasing the incidence of skeletal disorders in broiler chickens (Thiruvenkadan *et al.*, 2011). Similarly, environmental considerations can be included in BO and index calculations (van Arendonk, 2011).

Some breeders have also incorporated behavioural traits into their selection criteria. It may be considered unethical to select for behaviours that better suit an animal to an agricultural production environment, with some advocating that the production environment should be modified to suit the animal. However, it should be recognized that livestock populations have been selected for behavioural traits since their domestication. While altering the environment might be appropriate in some cases, such a change needs to be considered in the context of undesired negative impacts on other components of sustainability. It is conceivable that selection to better suit a population of animals to their production environment could improve animal welfare and productivity, thereby working towards multiple sustainability goals.

An example from the poultry industry illustrates this point well. Muir (1996) performed a selection experiment with a line of White Leghorns to improve adaptability and well-being of layers in large multiple-bird cages. Feather and vent pecking, and sometimes cannibalism can occur in multi-pen cages, a problem that can be managed with beak trimming young birds. Muir used a selection approach termed ‘group selection’. In this experiment, offspring from select roosters were housed as a group in multiple-bird cages, and the group was either selected or rejected based on the productivity of the group. An unselected control, with approximately the same number of breeders as the selected line, was maintained for comparison and housed in single-bird cages. After six generations, annual mortality of the selected line in multiple-bird cages decreased from 68% to 8.8%. Mortality of the selected line in multiple-bird cages was similar to that of the unselected control in one-bird cages. Annual survival improved from 169 to 348 days, eggs per hen per day rose from 52% to 68%, total eggs per hen from 91 to 237 eggs, and total egg mass from 5.1 to 13.4 kg, while average egg weight remained unchanged. The author concluded that these data suggested group selection could eliminate the need to beak-trim to avoid cannibalism by breeding hens to better suit multiple-bird cage production systems. These outcomes from group selection would seem to align with several sustainability goals, including decreased cannibalism and a resultant improvement in animal welfare, better production efficiency, reduced need for beak trimming, and group housing rather than single-bird cages. In reality it is likely that moving to alternative production systems (e.g. free range) will necessitate ‘reselecting’ for animals that are better suited to the new production environment. With too much space, birds become territorial and it is not uncommon to have much greater mortality in floor pens than cages due to the increased area (Muir and Cheng, 2004).

Other aspects of sustainability are not so easily addressed using a selection index approach. In an excellent review, the current status of valuing animal welfare issues in breeding goals is discussed in Nielsen *et al.* (2011). The difficulty of defining sustainable breeding goals is to value intangibles in monetary terms. Assigning MV to traits related to animal welfare continues to be a challenge because it is both a ‘public good’ and subject to ‘market failure’. Public good is an economic term meaning that the good is ‘non-rival’ and ‘non-excludable’ in consumption. Non-rival means that the consumption of the good by one individual (e.g. improved chicken housing) does not preclude another from consuming the same good. Non-excludable means that no one can be effectively excluded from using the good including those that may not eat chicken. Market failure is an economic term that refers to a situation

where, in any given market, the quantity of a product demanded by consumers does not equate to supply. For example, the public may overwhelmingly support the concept of cage-free poultry production, but the market does not respond because many consumers are unwilling to pay the additional costs associated with this production system. Studies show that market prices do not fully reflect the value people place on animal welfare. In general, peoples' opinions, values, beliefs and concerns with public interest may reflect a desire for a change of agricultural practices, but most 'consumers' (reflecting people's own interest in the market) only incidentally buy environmental and animal friendly products. The use of market prices as a sole measure of value will therefore underestimate the interest in improved animal welfare. As with MV, values placed on improvements in animal welfare will vary among different markets and are likely to be higher in locations with food sufficiency than in those without.

One approach to addressing this problem is to add 'non-market' value (NMV) to the selection index to consider non-monetary value placed on animal welfare or environmental concerns (Olesen *et al.*, 2000). In that case, the breeding goal (H) is a function of both MV and NMV values:

$$H = (MV_1\,EBV_1 + \ldots + MV_n\,EBV_n) + (NMV_1\,EBV_1 + \ldots + NMV_n\,EBV_n).$$

The principle behind this approach is that when defining BO for sustainable production, the selection response should be based on MV value, and then a NMV added for traits with unacceptable selection responses. While MV can be derived mathematically based on the contributions of traits to profit, NMV can be derived using methods such as stated preference techniques (consumers choose between products with various quality traits) and choice experiments (which are based on individual's willingness to pay for a specific product). Such approaches require decisions to be made about whether farmers and/or consumers should bear the NMV costs of improving traits related to animal welfare and other social concerns, or whether society should cover some of the losses, e.g. through regulations and or taxes in the form of government subsidized breeding programmes (Nielsen *et al.*, 2011). Without subsidies, the incentives for breeding companies to produce animals with improved animal welfare will be low. In the absence of a new market driver to incentivize the adoption of breeding for improved welfare, companies who focus on low heritability welfare traits may lose market share to companies that focus on high heritability production traits with a high MV.

In selection index theory, the MV reflects future prices and costs, about which there is inevitable uncertainty. However, many functional traits are valuable under all future prices and so their inclusion in BO is likely to be beneficial, both from an economic and animal welfare perspective. What is less certain is what relative emphasis should be placed on different social and animal welfare traits as societal norms are unpredictable and can change over time. The term future market value (FMV) is used to describe a value that is not linked to current market forces. Breeding companies may incorporate traits with a FMV into their BO in anticipation that consumers will demand a certain level of animal welfare in the future. Examples include a cattle breeding company GENO that proactively put selection emphasis on health and fertility traits of Norwegian red dairy cattle many years ago, and sheep breeders in Australia and New Zealand who include resistance to internal parasites in their BO (Nielsen *et al.*, 2011). However, this decision has direct negative economic consequences as placing greater emphasis on traits that do not contribute to profit at the current time slows genetic progress in traits that do, such as production traits. An indirect economic consequence of including additional traits in an FMV is higher recording costs to get accurate phenotypes for the additional traits in order to allow selection to be performed on them.

If the livestock sector is to meet the desire of the world's population for animal protein produced in a sustainable manner while ensuring food safety, animal welfare and the maintenance of rare and specialist breeds, the rate of genetic gain is going to have to accelerate markedly. There are a number of technologies that can be used to modify the variables that contribute to the selection response. In the following section, each of these components will be discussed in turn, with specific examples to emphasize the interplay between animal breeding and sustainability.

Accuracy of Selection

The development of statistical methods for accurately calculating EBV has been one of the major success stories of animal breeding. A relatively new DNA-enabled approach to calculate EBV of animals is genomic selection (GS), which was proposed by Meuwissen *et al.* (2001). This concept is based on the idea that sections of DNA associated with trait variation can be tracked by DNA markers spread throughout the genome and that this can be used to accurately predict the EBV of an animal at a young age. The approach essentially involves simultaneous selection on tens or hundreds of thousands of markers that are distributed throughout an animal's genome based on their relative effect on specific traits. These marker effects are estimated using the phenotypes and genotypes of a large number of animals from a reference population. Using this information, an animal in another population can have its genomic BV estimated using only genotypic information, before phenotypes are recorded.

This technology has already been adopted by the dairy industry to increase the accuracy of EBV on young bull sires prior to progeny testing, thereby enabling the inclusion of these young males in breeding programmes earlier in their lives, thus decreasing the generation interval. It is hoped that this technology will be able to contribute to sustainable animal breeding by providing previously absent selection criteria for traits of importance to sustainability goals. Many of these traits have previously been omitted from BO due to the expense and/or difficulty of phenotyping; using GS, EBV become available for a wide variety of traits once the animal has been genotyped. It is also thought that using GS may help to decrease rates of inbreeding per generation, because selection using this approach increases the emphasis on the Mendelian sampling of genes an individual receives from its parents (within family selection), as distinct from emphasizing the parent average as in traditional BLUP, which tends to select entire families (Daetwyler *et al.*, 2007). For example, two full siblings share the same parents, and therefore, before being phenotyped themselves, they would be expected to have similar genetic value. But if one of their parents was heterozygotic for a gene affecting a particular trait, one sibling may have received the beneficial allele, and the other sibling may have received the detrimental allele. In this case, GS would predict that the former sibling would have a higher BV than the latter sibling based on their different genotypes for that gene, while parent average would predict them to have the same value. Therefore, a breeder using GS could select between these full siblings while a breeder using traditional, pedigree-based selection would not be able to tell them apart until their phenotypes were recorded.

Selection Intensity

Perhaps no other technology has had as great an impact on accelerating the rate of genetic gain through increasing the intensity of selection than has artificial insemination (AI). Artificial insemination technology was introduced into the dairy industry and commercialized in the USA during the late 1930s to early 1940s and achieved a domestic market of 15.5 million units in 2002. Seventy per cent of all dairy cows in the USA are bred using AI, as are virtually all turkeys and chickens. Artificial insemination allows the extensive use of high-accuracy, genetically superior sires and plays a major role in design of breeding programmes and dissemination of advanced genetics. Although AI is now used routinely in animal breeding and human medicine, it was initially viewed with scepticism. There was a fear that AI would lead to abnormalities, and influential cattle breeders were originally opposed to the concept, as they believed it would destroy their bull market (Foote, 2002). When independent, university research demonstrated that the technology could be used to provide superior bulls, control venereal disease and produce healthy calves, subsequent adoption was swift. To put the impact of the genetic improvement enabled by AI in a sustainability perspective, consider that advances in the genetics, nutrition and management of US dairy cows over the last century have resulted in a greater than fourfold increase in milk production per cow, and a threefold improvement in production efficiency (milk output per feed resource input; VandeHaar and St-Pierre, 2006). About half of this 369% increase in production efficiency is attributable to genetic improvement enabled by AI. As a

result, a much smaller population of dairy cows supplies the US market. The US dairy cattle population peaked in 1944 at an estimated 25.6 million animals with a total annual milk production of approximately 53.1 billion kg. In 1997, dairy cattle numbers had declined to 9.2 million animals and total annual production was estimated at 70.8 billion kg. The advent of frozen semen also dramatically curtailed the number of natural service dairy bulls on farms, which further lessened the inputs required to produce a unit of milk (Capper *et al.*, 2009).

There is a trade-off associated with the rapid dissemination of genetics through populations by AI, and that is a reduction in genetic diversity. A good example of the reproductive potential of an elite dairy bull comes from a bull named Elevation, born in 1965. He had over 80,000 daughters, 2.3 million granddaughters and 6.5 million great-granddaughters (VanRaden, 2007). Such extensive use of small numbers of sire families has reduced the genetic diversity of the Holstein population. Intense selection leads to rapid genetic improvement, but it also reduces the relative number of parents or the effective population size. Worldwide, estimates of effective population size in Holsteins range from 100 to 150, despite the fact there are more than 3.7 million Holstein cows enrolled in milk recording in the USA. Reduced genetic diversity can cause a reduction in mean phenotypic performance as a result of inbreeding depression. This term refers to the decrease in fitness and vigour that results from the breeding of related individuals. One of the primary concerns related to inbreeding is reduced reproduction and fertility. It has been observed that dairy cow fertility has been declining at 1% per annum for several decades. For example, daughter pregnancy rate, a measure of how quickly cows become pregnant after having a calf, declined from 33% to 23% over the period from 1960 to 2007. As with many considerations associated with sustainability, some balance needs to be reached between the inherent conflict of accelerating the rate of genetic gain by increasing the intensity of selection on superior lines of cattle, and minimizing the rate of inbreeding. Statistical methods have been developed that allow optimization of the long-term response to selection and restricted rates of inbreeding by selecting the best animals while minimizing the average relationship among the selected animals (Meuwissen, 1997).

Genetic Variation

Of the 18 mammalian species and 16 avian species commonly consumed for food, six mammals (cattle, buffalo, sheep, goats, pigs and horses) and four birds (chickens, ducks, turkey and geese) are widespread. One common feature of large-scale animal production systems is that they are based on a few breeds and so reduce, rather than increase, genetic diversity. As human populations have experienced growth and increased their demand for dietary animal protein, highly productive breeds have replaced local breeds throughout the world. This has led to concern about maintaining the genetic diversity present in low-production breeds as a source of genetic variation in traits of interest to future breeding programmes. Genetic improvement programmes must always conserve genetic diversity for future challenges, both as archived germplasm (such as frozen eggs and sperm) and as live animals (Blackburn, 2004). According to the FAO, 20% of the roughly 7600 breeds reported worldwide are at risk and 62 breeds became extinct within the first 6 years of this century (FAO, 2007). Breeding animals for conservation differs from breeding for production purposes in that the focus is not on making genetic progress in certain traits, but rather on maintaining genetic variation in breeds with a low population size. The availability of DNA marker information, used to inform GS for selective breeding, will also help inform optimum management programmes to maintain genetic diversity. Ironically, some of the techniques that have resulted in a reduction in genetic diversity in agricultural populations (such as AI, embryo transfer and cloning) offer opportunities to preserve endangered breeds through gene banks. Several authors have described approaches to select which animals should be cryopreserved to maximize the genetic variation stored (Caballero and Toro, 2002). The frozen storage of genetic material is not without problems, including suboptimal fertility/viability due to freezing and thawing and the continued environmental degradation or population pressures that resulted in the breed becoming endangered in the first place.

Another approach to increasing genetic diversity is to introduce new variability using genetic engineering (GE). GE refers to the process of introducing recombinant DNA (rDNA)

into the genome of an animal such that the rDNA modification is stably transmitted to their offspring in a Mendelian fashion. Traditional animal breeding methods are typically used for the propagation of the transgene once the founder animal has been produced. The use of GE is most appealing when the allele substitution effect is very large resulting in a profound change in phenotype that would be difficult, if not impossible, to achieve using traditional breeding approaches (e.g. expressing a protein not found in that species). This approach can include gene addition (i.e. transgenesis), or targeted gene editing of the endogenous genome. The latter enables precise changes to be made at a specific location in the genome (e.g. creating a gene knock-out) without any other changes to the genome of an animal (i.e. without selection markers, or even the genome-wide changes caused by crossbreeding). This approach, which is being enabled by several new molecular methods e.g. zinc finger nucleases, and transcription activator-like effector nucleases (TALENs), can be used to supplement or replace a target allele present in one population with a preferred allele with known effect from another population (Fahrenkrug *et al.*, 2010).

Whether GE livestock fit in with sustainability goals will be greatly dependent upon the BO and production system. However, some GE livestock applications (e.g. disease resistance) would seem to align with many sustainability goals, such as improving animal well-being. Infectious diseases have major negative effects on poultry and livestock production, both in terms of economics and animal welfare. The costs of disease are estimated to be 35–50% of turnover in developing countries and 17% in the developed world. Improving animal health using GE has the added benefit of reducing the need for veterinary interventions and the use of antibiotics and other medicinal treatments. Efforts are underway to generate trypanosome resistance in cattle, which is a major problem for beef and dairy population in East Africa (Willyard, 2011). GE could also provide a humane method for sex selection in dairy and egg industries, where females provide the animal product (i.e. milk and eggs). Gene supplementation that feminizes male embryos (Smith *et al.*, 2009) or eliminates the production of male sperm in sires (Herrmann *et al.*, 1999) is technically feasible; the latter approach has the desirable outcome that the animals that are produced are not themselves GE (Fahrenkrug *et al.*, 2010). This change to sex-biased or sex-specific production of offspring would have the additional advantage of increasing overall efficiency of the production system (Hume *et al.*, 2011).

Similarly, the use of more productive GE animals should be given due consideration in the context of sustainability. Consider the controversial example of the AquAdvantage™ (AA) salmon, the first GE animal destined for the human food chain to attempt US regulatory approval (Van Eenennaam and Muir, 2011). The AA salmon is an Atlantic salmon carrying a Chinook salmon growth hormone gene controlled by an antifreeze protein promoter from a third species, the ocean pout. The mature weight of these fish remains the same as other farmed salmon, but their growth rate is increased by 400–600%, with a concomitant 25% decrease in feed input, decreased waste per unit of product and a shortened time to reach market weight (Du *et al.*, 1992). The AA salmon application that was reviewed by the US Food and Drug Administration (FDA, 2010) included management measures to abate the potential escape and interbreeding of GE fish by limiting the production to land-based tank culture facilities. The proposed locations were FDA-inspected and featured simultaneous, multiple and redundant physical and geographical containment measures. And as an extra precaution, additional levels of biological containment were proposed, including the production of 100% female fish (unable to interbreed) and triploidy induction (which results in sterility), with an average success rate of 99.8% (range 98.9–100%). The FDA review concluded that the likelihood that AA salmon could escape from confinement was considered very low. Despite these proposed containment measures, some groups maintain that any genetic risks associated with AA salmon are unacceptable. Absent from the debate over the AA salmon has been any discussion of the genetic implications of the escape of growth-selected lines of Atlantic salmon from conventional net pen aquaculture.

Atlantic salmon remains the most important farmed food fish in global trade. Since the mid-1980s, the yield of food fish from capture fisheries has been static at about 60 mMT year^{-1}.

Ocean-based net pen aquaculture has supplied the growth of the salmon supply since that time. It has been calculated that an extra 52 mMT of aquaculture production will be needed by 2025 if the current rate of fish consumption is to be maintained (Fletcher *et al.*, 2004). Selection for fast-growing fish using conventional breeding results in a shift in the allele-frequencies of many growth-associated genes. Farmed fish have been shown to have a fitness disadvantage, called a genetic load, in natural environments because domestication genes are only favourable in domesticated environments. Interbreeding between escaped farmed salmon and wild native fish can result in a 'substantial risk of extinction for natural populations' (Lynch and O'Hely, 2001). In principle, there is no difference between the types of concerns and potential magnitude of the environmental risks associated with the escape of growth enhanced GE salmon and those related to the annual escape of the millions of farmed Atlantic salmon that are genetically divergent from native populations in other ways, e.g. strains selected for enhanced growth (Schiermeier, 2003). Given these concerns a case could be made that raising GE salmon in contained land-based tanks is a more sustainable approach to salmon aquaculture than the ocean-based net pen aquaculture systems that currently supply over half of the world's salmon market.

Other examples of GE animals that have been developed for agricultural applications may also contribute to sustainability. Transgenic animals have been developed for disease resistance (Wall *et al.*, 2005) and environmental benefit. For example, the GE 'Enviropig' has decreased levels of phosphorus in its manure because it produces the enzyme phytase in its saliva, and is therefore able to metabolize dietary phytate (Golovan *et al.*, 2001). Given the large increase that is expected in both pig and poultry production in the developing world over the next 20 years as a result of the 'livestock revolution' (Delgado, 2003), decreasing the phosphorus levels in the manure of these monogastric species would likely have a huge worldwide environmental benefit. A number of researchers (Flint and Woolliams, 2008; Fahrenkrug *et al.*, 2010; Hume *et al.*, 2011; Niemann *et al.*, 2011) consider that GE animals 'can and will provide many of the solutions for tomorrow's agriculture' (Hume *et al.*, 2011). The current regulatory and political roadblocks to this technology are preventing its adoption in the developed world, but several developing countries are aggressively pursuing the development of GE animals to help provide a source of animal protein for their growing populations.

Generation Interval

We are currently entering an era where DNA technology will likely expand the repertoire of traits that can be addressed by breeding (Flint and Woolliams, 2008). Because GS offers an opportunity to improve the accuracy of EBV of young animals, it provides an approach to decrease the generation interval. It is for this reason that GS has experienced such widespread and rapid adoption in the dairy breeding sector where the need to determine the BV of a young bull typically involves waiting for milk production records from his daughters. This increases the male generation interval considerably compared with a BV estimate that can be obtained from a DNA test of a newly born calf. For example, in dairy populations the rate of genetic improvement is expected to double with the application of GS (Hayes *et al.*, 2009a).

GS may also offer selection criteria for traits that are not currently considered in BO due to an absence of objective, quantifiable measures upon which to base selection decisions. The GS approach is clearly attractive for difficult to measure traits such as reproductive success and longevity in varied environments, efficiency of nutrient utilization, animal temperament, stress susceptibility, innate resistance or susceptibility to disease, adaptability, and reduced GHG emissions. Hayes *et al.* (2009b) demonstrated how GS could be used to breed cattle better adapted to an environment altered by climate change. Preliminary results from the poultry industry suggest that GS focused on leg health in broilers and liveability or viability in layers can rapidly and effectively improve animal welfare (Cheng, 2010). In theory, GS offers the opportunity to provide DNA-based selection criteria for multiple sustainability traits simultaneously.

Considerable investment in both genotyping and phenotyping will be required to develop

the large phenotyped and genotyped reference populations that will be required to realize the full potential of GS. This strengthens the case for government and industry investment in GS initiatives (Hume *et al.*, 2011). For example, it might be possible to measure methane production on large numbers of cattle or sheep in a research herd in order to calibrate the GS prediction equation for wider application by breeders. Initial studies suggest there is a positive relationship between efficiency and reduced methane emissions (Zhou *et al.*, 2009). For disease surveillance, it might be possible to integrate GS approaches using the phenotypes that are collected as part of routine government-funded disease surveillance (e.g. collection of case-control samples from mortalities on a given farm). These approaches may ultimately reveal a sub-set of markers of sufficiently large effect that they can be cost-effectively combined into a DNA test with a reduced number of markers. This would decrease the cost per test, although the accuracy of trait prediction would likely also be decreased. In the future, selection on genetic markers is likely to become more common. As with other selection criteria, DNA-informed selection decisions should be based on the overall effect of the different genotypes on maximizing the selection response per unit time.

Although on the surface GS may arouse less public opposition because it utilizes naturally occurring genetic variation, some associated applications to reduce generation interval that are enabled by GS may be seen as contrary to animal welfare. These include the use of germ line approaches to shorten the generation interval, such as the harvest of oocytes from calves that are still *in utero* (Georges and Massey, 1991), or an approach where breeding is essentially done in the laboratory using GS to predict the EBV of cells derived from *in vitro* meiosis events (Haley and Visscher, 1998). Such animal breeding scenarios are largely hypothetical, but analogous manipulations in the world of plant breeding have met with great success. *In vitro* sexual recombination in combination with GS could rapidly accelerate the rate of genetic progress, and may also serve as a new way of generating genetic diversity (Hume *et al.*, 2011). As with all of the genetic technologies discussed in this chapter, there is a need to weight the use of technologies or genetic resources that accelerate the rate of genetic gain against any potential negative impact such use may have on competing sustainability goals.

Sustainable BO need to include both production and functional traits including disease resistance, reproduction, longevity and well-being to optimize the result of genetic improvement programmes. Genomic technologies may offer new opportunities to expand the phenotypes available as selection criteria for functional and other novel traits (e.g. feed efficiency, methane production) thereby enabling their inclusion in future BO. In addition, BO need to also account for the NMV associated with environmental, genetic diversity, ethical and social considerations. The weighting given to these different considerations will necessarily vary between countries and regions given the differing production environments, local cultural and social conditions, food security concerns and uncertainty about future circumstances. Having regionally appropriate BO will help to maintain genetic diversity among breeding stocks throughout the world. Ultimately, developing sustainable BO will require a nuanced balance among often conflicting animal welfare aspirations, social and environmental concerns, food safety, public health, genetic variability and production demands.

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6 Minimizing Environmental Impacts of Livestock Production Using Diet Optimization Models

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Introduction

The environmental impacts of agriculture have received great attention in the last decade. Several greenhouse gas (GHG) inventories have been developed worldwide, and mineral and nitrogen excretion from livestock is now subjected to governmental regulation in many countries (Oenema, 2004). In a recent report, nitrate levels in drinking water were examined in the Tulare Lake Basin and Monterey County portion of the Salinas Valley in California (Harter *et al*., 2012). The region is responsible for 40% of California's irrigated croplands and over half of California's dairy herds. This region also supplies water for 2.6 million people, which rely on groundwater for drinking purposes. The major source of human-generated nitrate was croplands (96%), with the two main sources being synthetic fertilizer (54%) and manure applications (33%). Establishing a tax on nitrogen fertilizers, including manure, was one of the mitigation strategies proposed in the report, prepared under contract with the State Water Resources Control Board (Harter *et al*., 2012). Tax-based policies have been implemented worldwide to act as mitigation strategies in various economic sectors. For instance, in the Netherlands, tax-based penalties were established with the former Mineral Accounting system, in which producers were taxed for positive balances of nitrogen and phosphorus (Oenema, 2004).

The two principal categories of environmental policies, in the regulation of gas emissions, are taxes and limitations on the amounts emitted by a firm (Weitzman, 1974). An extension of these two policy strategies is the cap and trade policy scheme, in which total emissions are capped but entities involved in the regulatory process are allowed to trade residual credits or permits. A cap and trade policy scheme has been implemented in US power plants, in the attempt to reduce mono-nitrogen and sulfur oxides. In the Netherlands, a cap and trade system was used to reduce nitrogen and phosphorus excretion by livestock, under the former Mineral Transfer Agreement System (Oenema, 2004). In contexts such as this one, tax-based policies and their derivations are options that can be adopted not only to decrease nitrate water levels but also to reduce other potential environmental impacts of livestock production.

The management of a livestock production unit requires that factors affecting the system, such as profitability, animal welfare and sustainability, be examined. Decisions are then made based on the contribution of individual factors

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to a common objective. Traditionally, mathematical models have been used to assist the system manager in making optimal decisions (DeLorenzo *et al.*, 1992; Tozer and Stokes, 2001; Cabrera, 2010). Models are mathematical representations of a system, which characterize the relationships between system elements and the producer's objective. In order to represent the production system fully, elements affecting the system should be described mathematically, and the relationships between model variables should be specified. The structuring of the model depends on the objective of the producer, on the knowledge of the system manager and on the data available. Data and information available for model development can be obtained from production unit records and the literature. Using the data collected and models developed in previous studies is essential in the integration of knowledge about different elements of a production unit. For example, if the objective is the development of a model that maximizes dairy profit, the model requires information about production costs, milk prices, animal production and regulatory policies. Environmental policies can restrict the system operation by imposing regulatory constraints, as discussed above. Therefore, the mathematical representation of regulatory policies relies on the characterization of environmental impacts from the system and on the establishment of relationships between animal production and the impacts generated. Consequently, examining the effects of the regulation of environmental impacts on the milk production system requires the establishment of relationships between dietary composition, milk production, GHG emissions and mineral and nitrogen excretion. In order to establish these relationships, several equations to predict methane emissions have been developed in the past two decades (Mills *et al.*, 2003; Ellis *et al.*, 2007) and previously existing models have been extensively evaluated (e.g. Wilkerson *et al.*, 1995; Ellis *et al.*, 2010). Mineral and nitrogen excretion models have been developed (James *et al.*, 1999; Nennich *et al.*, 2005) to predict excreted amounts and to examine effects of animal and dietary characteristics related to efficiency of nutrient utilization.

Furthermore, GHG inventories (IPCC, 2006; EPA, 2011), mechanistic models (Bannink *et al.*, 2011) and whole farm models have been developed (Chianese *et al.*, 2009) for the simulation and examination of feeding strategies and management practices that may reduce methane emissions. Most of these models can determine or predict reductions in methane emissions caused by dietary manipulation. However, with the exception of the model developed by Chianese *et al.* (2009), which includes a diet optimization sub-model, most models are not suitable for the determination of optimal dietary changes for a target reduction in methane emissions. Most importantly, most of these models cannot calculate the marginal costs of environmental impact mitigation under optimal decisions. The reductions in methane emissions with dietary changes are usually assessed with predetermined changes in dietary characteristics. These predetermined dietary changes are in essence heuristic and not based on mathematical techniques that can search a specified feasible region for the optimal combination of feeds that meet emission targets. The development of a diet optimization model, in which feed selection is sensitive to environmental impacts, can assist producers in formulating diets that will contribute to their meeting environmental policy demands. The marginal costs of mitigation strategies can be derived under optimal decisions, and producers can use sensitivity analysis when examining possibilities for achieving regulatory demands. In the optimization framework, policy makers and producers could *a priori* examine the effects of imposing a policy. For example, the effect of methane mitigation strategies on the demand for human-edible feeds, producer's marginal revenues and system sustainability can be examined before a regulatory policy is implemented.

In this chapter, we will present some of the optimization modelling techniques that can be used to reduce the environmental impacts of livestock and to assess the effects of environmental policies on the production system. Specifically, this chapter: (i) examines feed management practices that can reduce methane emissions and mineral excretion from livestock; (ii) reviews and describes mathematically the optimization modelling techniques frequently used for diet optimization and the minimization of livestock production environmental impacts; and (iii) presents an optimization model that can formulate diets when environmental policies are

enforced. The model incorporates the effects of policies on feed selection, profitability and environmental impacts for a hypothetical dairy cattle herd. Model results are discussed, and the marginal costs of methane emissions mitigation strategies are generated through a linear programming framework.

Feed Management

Feed management is a general term, and can include several areas in a production unit. Purchasing of feeds, feeding strategies, feed storage, mixing and formulation of diets are key elements in feed management. Information related to animal production also plays a key role in feed management, for example, animal characteristics that affect animal nutrient requirements. In this chapter, we discuss two feed management topics that can be applied to the minimization of the environmental impacts of livestock. The first topic is related to the incorporation of between animal variability into diet formulation and the use of this information in reducing nutrient excess feeding. The precise feeding concept is introduced, with an example of different feeding phases throughout the feeding period of pigs. The second topic discussed is related to practical aspects of diet optimization, e.g. the conversion of a system such as the NRC (2001), in which energy and protein availabilities are intake dependent, into a linear programming framework. The division of feeds based on digestible fractions, for different levels of intake and production, can increase precision in the calculation of feed nutrient availability and, theoretically, reduce the environmental impacts of livestock production.

The precision feeding concept can be incorporated in a production unit through different strategies. For example, increasing the number of feeding phases in a production system reduces the variability of animals within a pen. In the case of growing animals, weighing the animals frequently can increase the precision in calculating nutrient requirements. In essence, a better understanding of feed nutrient availability and a more precise determination of animal nutrient requirements are key elements in feeding diets that deliver nutrients at the requirement level. Pomar *et al.* (2009) examined precision feeding techniques applied to growing pigs, and from observations that nutrient requirements vary greatly between pigs with similar characteristics, different methods of estimating nutrient requirements were discussed. In a population of animals, between-animal variability plays a role in determining the proportion of animals that have nutrient requirements met with a diet formulated for the average animal. Under the assumption that nutrient requirements in a population of animals follow a normal distribution, half of the animals are being underfed and half of the animals are being fed excessive nutrients. The reasoning is that, in a normal distribution, the mean coincides with the median, which represents the 50th distribution percentile. By using the cumulative distribution of a specific nutrient requirement, we could determine the proportion of animals that are meeting their nutrient requirements under different feeding strategies, and investigate the relationship between nutrient excretion and diet formulation. As a result, the trade-off between increasing the proportion of animals that have their nutrient requirements met and increased nutrient excretion can be examined. For a comprehensive discussion of the mechanisms by which precise feeding strategies can increase the efficiency of nutrient utilization, refer to Pomar *et al.* (2009), where an application of this concept is described in the feeding of growing pigs.

A better understanding of feed nutrient composition and availability for animal production can be achieved by the division of feeds according to nutrient digestibility. Feeding systems in which protein fractions are divided according to digestibility (Sniffen *et al.*, 1992; NRC, 2001) enable, for instance, a more precise determination of protein supply. Accordingly, the division of feeds based on digestibility and availability leads to a more specific and precise diet formulation. Therefore, diets can be formulated to deliver nutrients at the calculated animal requirement level. On the other hand, the division of feeds according to nutrient digestible fractions is usually dependent on the animal production level and dry matter intake. Consequently, knowledge of animal intake is necessary for the determination of feed nutrient availability and animal intake should be determined before the diet is formulated. From a diet formulation point of view, especially from a linear programming perspective, fixing the intake

at a certain level greatly reduces the model flexibility in finding combinations of feeds that meet animal nutrient requirements at a minimum cost. Also, in a linear programming diet optimization model, feed intake is the sum of the elements in the solution vector, and it is usually not known before the diet is formulated. But if the intake is not fixed at a determined level, the model feed composition matrix changes for distinct solution vectors and so does the nutrient supply, because nutrient availability is dependent on the intake level. In order to circumvent this problem, an iterative algorithm can be structured in which an initial feed composition matrix can be created with the use of an initial estimate of the dry matter intake and the model can then be solved. The feed composition matrix can be updated through the calculation of the composition of feeds with the model solution intake. The model can then be re-solved with the use of the updated feed composition matrix (Moraes *et al.*, 2012). Through the iterative updating of the model feed composition matrix and solving of the model, differences in the feed composition matrix can be made as small as desired by increasing the number of iterations. Therefore, a system with nutrient fractions divided according to digestibility can be used in a linear programming framework, and diet formulation can be specific to the model determined level of intake.

Finally, from a feed management perspective, probably the most important aspect with respect to mineral excretion is the availability of mineral sources for diet formulation, especially the separation of mineral sources. It is conceptually easy to understand that increasing the number of feeds available for diet formulation increases the model's flexibility in finding a minimum cost diet. This concept has a direct application for supplying minerals to livestock. Traditionally, one mineral premix composed of the nutritionally required minerals is added to the diet in a quantity determined by the most limiting mineral. Besides the most limiting mineral, all other minerals were fed in surplus and the excessive amounts were excreted in the urine or faeces because most mineral intake above the requirement level is excreted in faeces and/or urine. This concept is being used to formulate diets for dairy cows in the California Central Valley. In order to reduce salt excretion by dairy cows, the sodium chloride, or the choice of sodium source, is now available separately from the mineral premix and the amount of salt fed is determined by the sodium requirement, thus reducing sodium excretion. The concept can be applied to any mineral, or more generally, to any nutrient source that is competitively priced or environmentally regulated.

Feed management practices are key elements in a livestock production unit operation. A better understanding of feed nutrient composition and animal nutrient requirements can lead to a more precise diet formulation, reducing dietary costs and excessive animal mineral and nitrogen excretion. Precision feeding concepts can be used in systems in which the implementation of these techniques is economically and practically feasible. Variability of nutrient requirements for a population of animals can be examined and feeding strategies can be examined from a probabilistic perspective. Feed nutrient composition can be characterized according to nutrient availability at different levels of animal productivity and intake. And, finally, the feeds selected for diet optimization, for example mineral sources, can have a great impact on the amounts of nutrients fed and excreted.

Optimization Techniques Applied to Diet Formulation

Modelling optimization techniques have been extensively used in agriculture since the early 1960s (Dent, 1964; Black and Hlubik, 1980; Kennedy, 1986). Dynamic programming and linear programming are the most frequently used techniques in agricultural systems management. The objective of this section is the provision of a mathematical description of these techniques and applications to diet formulation and to the minimization of environmental impacts from livestock. Dynamic programming is introduced, and linear programming and extensions of the simplex algorithm, such as stochastic programming and multi-criteria programming, are examined. The last part of this section describes strategies for implementing regulatory policies in the diet formulation model and introduces an example of model-driven methane mitigation strategies in a dairy herd.

Dynamic programming

Planning and resource allocation are traditional applications of dynamic programming in forestry and fisheries management (Kennedy, 1986). A classical application in livestock science is the animal replacement problem. Several dynamic programming and Markov decision models (Kristensen, 1991; Kristensen and Søllested, 2004) have been used to find optimal replacement policies in animals of different species. Dynamic programming applications in feed formulation are not as widely used as linear programming applications; however, the strength of dynamic programming is that decisions can be made over time. Optimum combinations of feeds can be identified and resources that optimize some utility function, as adopted policies evolve over time, can be characterized. An application of dynamic programming in diet optimization is the identification of a sequence of optimum feeding strategies in relation to distinct production rates. The example from Kennedy (1986) illustrates the optimal fattening of a steer over time. The problem is based on the choice between sequences of rations that will produce different daily gains at different costs when fed to animals of various body weights. In this example from Kennedy (1986), it is assumed that the six daily gains range from 0.25 to 1.5 kg at intervals of 0.25 kg. The length of each stage, or the time between decisions, is assumed to be 28 days and the initial steer body weight is 300 kg. It is a four-stage scenario, in which at the last stage steers are worth US\$3 kg^{-1} live weight. For specific diet costs and rates of gain, refer to Kennedy (1986). The maximum body weight specified was 440 kg, and the problem then becomes the identification of the sequence of diets that maximize the net revenue. The model can be solved by backward induction techniques and be mathematically represented by the recursive equation:

$$\begin{aligned} V_i\{x_i\} &= \max[-a\{x_i,\ u_i\} \\ &\quad + V_{i+1}\{x_i + 28u_i\}], \\ &\quad \text{for } i = 3, 2, 1 \end{aligned}$$

with

$$V_4\{x_4\} = 3x_4, \quad \text{for } i = 4 \tag{6.1}$$

Subject to

$$\begin{aligned} & u_i \in (0.25,\ 0.50, \ldots,\ 1.50) \\ & x_1 = 300 \\ & x_i + 28u_i \le 440 \end{aligned}$$

where $V_i\{x_i\}$ is the value derived by having daily gain u_i given that the body weight is x_i, $a_i\{x_i, u_i\}$ is the dietary cost for live weight x_i and daily gain u_i at the ith stage.

Extensions of this simple dynamic problem are presented by Kennedy (1986) and Glen (1980), including maximizing returns per unit of time and the use of an animal replacement model in conjunction with this feeding rate model. The objective function in the dynamic program is constructed based on the objective of the optimization. The flexibility in specifying the model is tremendous, enabling the development of a large variety of modelling structures applied to the minimization of diet costs and livestock environmental impacts. The minimization or maximization of various types of objective functions can be performed; the restriction in the technique use relies on the ability of the decision maker in describing the problem mathematically. For instance, the determination of optimal policies in the dynamic programming model can be hampered by the difficulty in the definition of the model final conditions (Soetaert and Herman, 2009). Moreover, realistic representations of the model state space may have very large dimensions, exposing the modeller to a complex problem, often referred to as the curse of dimensionality (Kristensen *et al*., 2008). As a result, if the behaviour of state variables is confidently known over time, other modelling techniques may represent simpler strategies, for example linear programming.

Linear programming

Linear programming is the most widely used optimization technique in diet formulation. Since the development of the simplex algorithm by Dantzig (1963), several extensions of this method have been adapted to the diet formulation problem. In a brief mathematical description, linear programming can be characterized in terms of an objective function, which describes the contributions of each decision variable to the optimal value, and constraints, which limit the use of scarce resources. For linear programming assumptions and an extensive mathematical and economical description, refer to Winston (1987).

Briefly, the diet formulation linear programme can be represented algebraically as:

$$\begin{aligned} &\min \sum_{j=1}^{n} c_j x_j \\ &\text{Subject to} \\ &\sum_{j=1}^{n} a_{ij} x_j \geq, \leq b_i, \quad \text{for } i = 1,\ 2,\ \ldots,\ m \\ &x_j \geq 0,\quad j = 1,\ 2, \ldots,\ n \end{aligned} \tag{6.2}$$

where c_j is the cost of feed j, x_j is the amount of feed j, a_{ij} is the content of nutrient i in feed j, b_i is the animal's requirement for nutrient i, n represents the number of feeds and m represents the number of constraints. The constraints define a feasible region S that must contain the solution. Other constraints may be included to define the feasible region S.

The control in formulating the objective function and constraints allows great flexibility in specifying the objective of optimization and the feasible region S. In the linear programming framework, results are assessed with the vector of solutions and the level of scarce resources usage (left side of constraint equations). Sensitivity analyses are used to examine the marginal costs of restricting resource usage and ranges for which model solutions are invariant. Shadow prices represent the change in the objective function optimal value when a constraint equation is relaxed or strengthened. Therefore, they represent the marginal cost of strengthening or relaxing a constraint by one unit. The classical application of the linear programming model to diet optimization is the formulation of minimum cost and maximum profit diets. Recently, extensions of the least-cost diet formulation model were developed to minimize the environmental impacts of livestock (Jean dit Bailleul *et al.*, 2001; Pomar *et al.*, 2007; Moraes *et al.*, 2012). For instance, by including an environmental term in the traditional linear programming algorithm, Pomar *et al.* (2007) formulated diets with reduced levels of phosphorus, consequently reducing phosphorus excretion by pigs. Similarly, Jean dit Bailleul *et al.* (2001) modified the traditional least-cost formulation algorithm to decrease nitrogen excretion by pigs. In both studies, the objective function included a term representing the cost associated with the calculated nutrient excess and unavailable fractions. The model proposed by Pomar *et al.* (2007) applies the same concept as the model proposed by Jean dit Bailleul *et al.* (2001), and it can be described mathematically as (Pomar *et al.*, 2007):

$$\begin{aligned} &\min \sum_{j=1}^{n} \left(c_j + \beta r_j\right) x_j \\ &\text{Subject to} \\ &\mathbf{x} \in S \end{aligned} \tag{6.3}$$

where c_j is the cost of feed j, r_j is the amount of phosphorus resulting from the jth feed, x_j is the amount of feed j, β is the unit cost associated with the excess and unavailable phosphorus fraction, $\mathbf{x}$ is the vector of solutions and S is the feasible region, as described in Eqn 6.2. Values of β can be viewed as taxes for phosphorus excretion.

Phosphorus intake decreased with increasing β's, as expected, and diet cost increased with higher β values. The marginal reduction of phosphorus was calculated from the ratio between the change in total phosphorus and ingredient cost at critical values of β, i.e. values of β where the solution vector changed basis. When the methodology was applied to Canadian data (Pomar *et al.*, 2007) during June 2002 in Quebec, the first change in solution basis was achieved at a β equal to US\$0.00696 kg^{-1}. Phosphorus dietary content decreased from 5.287 to 5.181 g kg^{-1} and dietary costs increased from US\$196.1 to US\$196.8 t^{-1}. For this critical value of β, the marginal reduction of P excretion was 143 g US\$$^{-1}$ (Pomar *et al.*, 2007).

Stochastic programming

The linear programming diet formulation model can be extended through the inclusion of uncertainty, to the composition of feeds, to the animal nutrient requirements or to the feed costs. For example, if we assume that feed costs are not known with certainty over time, which is a reasonable assumption, they become stochastic elements in our model. By assuming that they vary according to some probability distribution function, we specify some degree of certainty about the values of the vector $\{c_j\}$. We can, therefore, construct a stochastic programming model to determine diets that have minimum variance in feed costs under a reference maximum expected cost. Let $\mathbf{c}$ represent the $\{c_j\}$ vector, and assume that $\mathbf{c} \sim \mathrm{N}(\mu_{\mathbf{c}}, \Omega_{\mathbf{c}})$, where $\mu_{\mathbf{c}}$ is the expected value

of $\mathbf{c}$ and $\Omega_\mathbf{c}$ is the variance-covariance matrix of $\mathbf{c}$. It can be easily shown that the expected value of $\mathbf{c}^\mathbf{T}\mathbf{x}$ is equal to $\mu_\mathbf{c}{}^\mathbf{T}\mathbf{x}$ and that the variance of $\mathbf{c}^\mathbf{T}\mathbf{x}$ is equal to $\mathbf{x}^\mathbf{T}\Omega_\mathbf{c}\mathbf{x}$, where superscript $\mathbf{T}$ represents the transpose of the vector. The stochastic programming model, in which $\mathbf{c}$ is a random vector, can then be represented, in matrix notation, as:

$$
\begin{aligned}
&\min \mathbf{x}^{\mathrm{T}}\Omega_{\mathrm{c}}\mathbf{x} \\
&\text{Subject to} \\
&\quad \mu_{\mathrm{c}}{}^{\mathrm{T}}\mathbf{x} \le e \\
&\quad \mathbf{x} \in S
\end{aligned} \tag{6.4}
$$

where $\mathbf{x}$ is the vector of decision variables, $\mathbf{x}^\mathbf{T}\Omega_\mathbf{c}\mathbf{x}$ is the variance of $\mathbf{c}^\mathbf{T}\mathbf{x}$, $\mu_\mathbf{c}{}^\mathbf{T}\mathbf{x}$ is the expected value of $\mathbf{c}^\mathbf{T}\mathbf{x}$, e is the reference value chosen for the maximum expected diet cost and S is the feasible region, as in Eqn 6.2.

Various objective functions can be formulated to accommodate distinct optimization objectives (Shapiro *et al.*, 2009). For example, the risk someone is willing to take can be included in the objective function, and a stochastic programming model can be specified based on the determined risk (Hazell and Norton, 1986). Stochastic programming models have been traditionally applied in diet formulation to incorporate uncertainty in the feed composition matrix, through the use of chance constraints (Chen, 1973; St-Pierre and Harvey, 1986a, b). The idea behind chance-constrained programming is the formulation of a model in which a determined constraint is met at a certain probability level. An application to the diet formulation problem would be the formulation of a constraint in which the requirement of a determined nutrient is met with certain probability. In the context of a least-cost diet, a model with probabilistic constraints can be described algebraically as:

$$
\begin{aligned}
&\min \sum_{j=1}^{n} c_j x_j \\
&\text{Subject to} \\
&\quad \Pr\left(\sum_{j=1}^{n} a_{ij} x_j \ge b_i \right) \ge \alpha_i \\
&\quad \mathbf{x} \in S
\end{aligned} \tag{6.5}
$$

where c_j is the cost of feed j, x_j is the amount of feed j, a_{ij} is the content of nutrient i in feed j, b_i is the animal requirement of nutrient i, α_i is the probability of meeting the ith nutrient requirement, $\mathbf{x}$ is the vector of feeds and S is the feasible region. The probabilistic constraint, along with other nutrient constraints, defines the feasible region S.

In the case where the nutrient requirement (b_i) is the random variable, a certain degree of certainty can be specified to the random nutrient requirement through the specification of a probability distribution function. The probabilistic constraint can then be formulated to incorporate the uncertainty into a parametric form. Using probability theory (Roussas, 2003), the chance-constrained equivalent to Eqn 6.5 can be represented as:

$$
\begin{aligned}
&\min \sum_{j=1}^{n} c_j x_j \\
&\text{Subject to} \\
&\quad \sum_{j=1}^{n} a_{ij} x_j \ge F_{b_i}^{-1}\,(\alpha_i) \\
&\quad \mathbf{x} \in S
\end{aligned} \tag{6.6}
$$

where c_j is the cost of feed j, x_j is the amount of feed j, a_{ij} is the content of nutrient i in feed j, F_{b_i} is the distribution function of the random variable b_i, α_i is the probability of meeting the ith nutrient requirement, $\mathbf{x}$ is the vector of solutions and S is the feasible region. The chance constraint, along with other nutrient constraints, defines the feasible region S.

The probability distribution function assigned to the random variable in the chance constraint is determined by the random variable distribution. For instance, assume that the requirement of the ith nutrient in a population of animals follows a normal distribution, with expected value μ_{bi} and variance σ^2_{bi}, i.e. $b_i \sim \mathrm{N}(\mu_{bi}, \mu^2_{bi})$. Then, our knowledge about the nutrient requirement can specified through the probability distribution of a normal distribution, and our random variable b_i can be standardized into a standard normal random variable. In this framework, the chance-constrained programming model can be represented mathematically as:

$$
\begin{aligned}
&\min \sum_{j=1}^{n} c_j x_j \\
&\text{Subject to} \\
&\quad \sum_{j=1}^{n} a_{ij} x_j \ge \mu_{bi} + k_{\alpha_i}\,\sigma_{b_i} \\
&\quad \mathbf{x} \in S
\end{aligned} \tag{6.7}
$$

where c_j is the cost of feed j, x_j is the amount of feed j, a_{ij} is the content of nutrient i in feed j, μ_{bi} is

the expected value of the requirement of nutrient i, k_{α_i} is the value of the standard normal variable when the probability of the constraint to be satisfied is α_i (and can be found in Z tables), σ_{bi} is the standard deviation of the requirement of the ith nutrient, $\mathbf{x}$ is the vector of solutions and S is the feasible region as described in Eqn 6.2. The chance constraint, along with other nutrient constraints, defines the feasible region S.

The development of a diet formulation model with probabilistic constraints determining uncertainty in the feed composition matrix follows the same concept of Eqn 6.6. The difference from the random nutrient requirements model is that now a_{ij} are the random variables. For an application of stochastic programming in diet formulation, in which single and joint chance constraints determined the inclusion of uncertainty in the feed composition matrix, refer to St-Pierre and Harvey (1986a, b). A deterministic equivalent chance constraint can usually be derived and used to identify optimal solutions in the stochastic programming framework (Charnes and Cooper, 1963; Symonds, 1968). However, stochastic programming models and their deterministic equivalents might exhibit non-linear formulation, increasing the complexity in the model optimization (Kall and Wallace, 1994). Linear approximations of chance constraints have been proposed in the literature (Olson and Swenest, 1987) and were investigated by St-Pierre and Harvey (1986a), in which non-linear chance constraints were specified to introduce uncertainty in feed composition. In the non-linear programming framework, specialized optimization algorithms are required in model optimization. The interpretation of solutions and sensitivity analysis are not straightforward, as in the linear programming, for example the existence of local versus global optimum. A valuable element in the examination of optimum scenarios is the evaluation of dual values, especially shadow prices. In the non-linear programming framework, the calculation and interpretation of shadow prices are more complex than in linear programming models. From an environmental perspective, shadow prices have valuable information because they can be interpreted as the marginal costs of mitigation strategies and can be used, in an economic framework, to derive abatement cost curves. Therefore, the inclusion of uncertainty into the linear programming model might increase the complexity in the interpretation of results and, from and an environmental perspective, reduce the usefulness of sensitivity analysis. Nevertheless, in cases where solutions and shadow prices are derived with confidence, and can be directly interpreted, the inclusion of uncertainty can represent the variation in biological variables better. Moreover, probabilistic assessments can be incorporated into decision making. For instance, the trade-off between increasing the chance of meeting animal nutrient requirements and increasing environmental impacts can be examined.

Multi-criteria programming

The simplex algorithm can be modified to optimize a linear program when there is more than one objective function to be maximized or minimized. Multiple criteria programming usually involves the minimization of deviations or variations from specific goals, which are usually set by individual optimizations. From an environmental perspective, multi-criteria linear programming can be used to minimize livestock environmental impacts and diet costs. Recently, a multi-criteria programming model was developed to minimize diet costs and nitrogen and phosphorus excretion from growing pigs (Dubeau *et al.*, 2011). The model can be represented, in matrix notation, as:

$$\begin{cases} \min Z_1 = \mathbf{c}^{\mathrm{T}}\mathbf{x} \\ \min Z_2 = \mathbf{p}_{\mathrm{r}}{}^{\mathrm{T}}\mathbf{x} \\ \min Z_3 = \mathbf{p}_{\mathrm{h}}{}^{\mathrm{T}}\mathbf{x} \end{cases} \qquad (6.8)$$

$$\text{Subject to}$$
$$\mathbf{x} \in S$$

where $\mathbf{c}$ denotes the vector of feed prices, $\mathbf{x}$ is the vector of feeds, $\mathbf{p_r}$ is the vector of the nitrogen content of feeds, $\mathbf{p_h}$ is the vector of the phosphorus content of feeds and S denotes the feasible region, as described in Eqn 6.2.

In order to find the solution to this problem, efficient solutions that comprise the efficient set must be identified. A solution is efficient if there exists no other feasible solution $\mathbf{x}'$ such that $z_i(\mathbf{x}') \le z_i(\mathbf{x})$ for all i, and $z_j(\mathbf{x}') < z_j(\mathbf{x})$ for at least one j, where z denotes the objective function,

i and *j* denote the criteria or goal (Dubeau *et al.*, 2011). Efficient solutions can also be described through the idea of Pareto efficiency (Tamiz *et al.*, 1999), which characterizes a solution as efficient if and only if there is no other solution that improves the objective value of a determined goal without negatively affecting other goals. Results from the three criteria model (Eqn 6.8), when applied to the growing pig data from Quebec (Dubeau *et al.*, 2011), show that when a nitrogen excretion reduction of 25% and a phosphorus excretion reduction of 16% were achieved, the diet cost increased by 10%.

A classical application of multi-criteria programming is goal programming. The formulation of a goal-programming model involves a 'pay-off' matrix because optimization goals can conflict. In the goal-programming framework, each objective is optimized individually, and an objective function is created to minimize deviations from the result of the individual optimizations. The individual objective function of each goal is then turned into an equality constraint, which will restrict the system along with resource constraints (Romero and Rehman, 1989). The goal-programming model can be described mathematically as:

$$\begin{aligned}
&\min \sum_{j=1}^{n} w_j (n_j + p_j) \\
&\textit{Subject to} \\
&\quad z_j(\mathbf{x}) + n_j - p_j = g_j \\
&\quad \mathbf{x} \in S \\
&\quad n_j,\ p_j \geq 0
\end{aligned} \tag{6.9}$$

where w_j is the weight of goal *j*, n_j is the negative deviation from goal *j*, p_j is the positive deviation from goal *j*, $\mathbf{x}$ is the vector of decision variables, g_j is the objective target for the goal *j*, $z_j(\mathbf{x})$ is the objective function for the goal *j* and *S* is the feasible region as defined in Eqn 6.2.

Andre *et al.* (2009) developed goal-programming models capable of incorporating environmental and economic goals in policy development. A model application in Spain was presented, in which macroeconomic and environmental goals (gas emissions) were applied in an equilibrium model. The formulation of goal-programming models requires that a ranking be established between criteria or that weights, which assign a level of importance to each goal, be specified. However, the derivation of the weights is usually a complex task and can often be particular to each system to be optimized (Hannan, 1985). Moreover, the weight attribution for each deviation goal can be subjective and lead to distinct efficient solutions (Gass, 1987). Therefore, the use of multi-criteria programming is often limited to systems for which weights are specified with confidence or the most efficient solution vector can be identified graphically or numerically (Dubeau *et al.*, 2011).

Incorporation of environmental policies in the diet optimization model

The minimization of environmental impacts from livestock production units is traditionally enforced by the implementation of a regulatory policy. Therefore, the model structure should incorporate the effects of the policy in the optimization. The regulatory policy can be incorporated into the model through different strategies. Pomar *et al.* (2007) and Jean dit Bailleul *et al.* (2001) included an environmental term in the objective function to represent a tax on phosphorus and nitrogen excretion in pigs. In a recent publication, Moraes *et al.* (2012) modified a base traditional least-cost diet formulation model by using two different regulatory policy strategies. Methane emissions were taxed or capped and diets were formulated to minimize costs and methane emissions simultaneously. The marginal costs of methane emissions mitigation were derived using the shadow prices from a methane emissions constraint and a cap and trade policy scheme was simulated using the carbon credit market. The effects of these two policy strategies on feed selection for diet formulation, system profitability and sensitivity between environmental impacts were extensively examined. In the next section, the model by Moraes *et al.* (2012) is described in detail, and the results generated by the model structure are examined. The use of this modelling framework provides a comprehensive example of the utilization of optimization models in minimizing the environmental impacts of livestock in an optimal manner. The modelling framework also provides information for determining whether

policies regulating methane emissions are appropriate in dairy production systems.

Minimizing Diet Costs and Environmental Impacts in a Dairy Herd

Several optimization modelling techniques have been presented up to this point. A brief mathematical description and selected examples have also been provided. However, applications of these techniques in minimizing the environmental impacts of livestock production were briefly introduced, and the results of specific modelling strategies were briefly discussed. This section provides a comprehensive examination of a diet optimization model (Moraes *et al.*, 2012) implemented to minimize diet costs and methane emissions simultaneously from a dairy cattle herd. A mathematical description of the model is provided, and key results are discussed to demonstrate the role of the model structure in minimizing methane emissions from dairy cattle and examining the effects of regulatory policies in the milk production system.

A linear programming model was developed (Moraes *et al.*, 2012) to minimize methane emissions from a hypothetical dairy herd through the inclusion of environmental policies in the least-cost diet model framework. The hypothetical dairy cattle herd was composed of animals in seven categories: three categories of heifers, dry cows, primiparous and multiparous cows at early lactation and mid- to late lactation. Feed composition and animal nutrient requirements were calculated using the NRC (2001). The main idea behind model development was the construction of a mathematical tool to assist producers in meeting demands set by environmental policies and, at the same time, develop a model that could jointly minimize diet costs and the environmental impacts of dairy cattle. A linear programming model was the method of choice because it would provide a flexible framework for including the effects of environmental policies either in the model constraints or in the objective function. Multi-criteria programming, especially goal programming, was not the method of choice because, for different goal weights, a different solution was reached (as expected) and the subjectivity in assigning the weights overpowered the possibility of optimizing multiple objectives. The model is deterministic, i.e. there is no uncertainty in feed costs, in animal nutrient requirements, or in feed composition. A deterministic linear programming model was used because the shadow prices for a methane mitigation constraint were an important part of the model output; therefore, the model's linearity ensured that interpretation of results would be straightforward.

The model examined the effects of enforcing two categories of environmental policies on methane emissions from dairy cattle, thus forcing a reduction of methane emissions by two distinct strategies. The first policy strategy (the TAXM model) involved the taxation of methane emissions by using carbon dioxide equivalent prices in the carbon market. The model forced a reduction in methane emissions by changing the dietary composition according to a methane prediction equation. The tax on methane emissions is included through the objective function, and the reduction in methane emissions is, conceptually, achieved through the formulation of a new diet, at a higher cost, which has a lower proportion of fibre. The second strategy to reduce methane emissions through a regulatory policy (the REDM model) involved the reduction of emissions by predetermined amounts. The reduction was forced through a methane constraint which restricted emissions based on a baseline model (the BASEM model). The reduction in methane was also achieved through a higher cost diet, which usually had an increased proportion of protein and energy content and a decreased proportion of fibre. Nitrogen and mineral excretions were calculated through mass balance and were used to assess the sensitivity of excretions to reductions in methane emissions achieved through dietary manipulation. The formulation of TAXM can be described mathematically as:

$$
\begin{aligned}
&\min \sum_{a=1}^{7} \sum_{j=1}^{19} x_{ja} c_j + \sum_{a=1}^{7} e_a p \\
&\text{Subject to} \\
&\quad \mathbf{x} \in S
\end{aligned}
\tag{6.10}
$$

where x_{ja} is the amount of feed j for animal category a, c_j is the cost of feed j, e_a is the amount of methane emitted by animal category a, p is the

fixed cost (or tax) of methane emission, $\boldsymbol{x}$ is the vector of solutions and S is the feasible region. For a detailed description of S, refer to Moraes *et al.* (2012).

The formulation of the REDM model can be described mathematically as:

$$\min \sum_{a=1}^{7} \sum_{j=1}^{19} x_{ja} c_j$$

Subject to

$$\sum_{a=1}^{7} \sum_{j=1}^{19} x_{ja} p_1 NDF_j + \sum_{a=1}^{7} \sum_{j=1}^{19} x_{ja} p_2 ME_j \leq A_T CH_4 1 - Int \tag{6.11}$$

$$\mathbf{x} \in S$$

where x_{ja} is the amount of feed *j* for animal category *a*, c_j is the cost of feed *j*, p_1 and p_2 are the methane emission prediction equation parameters, NDF_j is the neutral detergent fibre content of feed *j*, ME_j is the metabolizable energy content of feed *j*, A_T is the predetermined reduction in methane emissions, $CH_4 1$ is the amount of methane emitted in the baseline scenario, *Int* is the intercept of the methane emission prediction equation representing the total herd, $\boldsymbol{x}$ is the vector of solutions and S is the feasible region. For a detailed description of S, refer to Moraes *et al.* (2012).

Equations 6.10 and 6.11 are complementary in the sense that, conceptually, a tax that achieves any desired mandated emissions reduction can be derived. Therefore, it is possible to establish a tax that leads to an exact predetermined reduction in methane emissions. In this context, the formulation of the REDM enabled shadow price calculations of the methane emissions constraint under different regulatory scenarios. Shadow prices can be interpreted as the marginal costs of reducing one unit of methane emissions through dietary manipulation. Therefore, the cost of mitigation strategies could be examined in relation to different policy intensities. Similarly, this model can be easily adapted to minimize any environmental impact that can be mitigated by dietary manipulation. For example, if a constraint equation is set to restrict the amount of nitrogen excreted by livestock, the marginal cost of reducing nitrogen excretion through dietary manipulation can be derived.

Results generated by the model structure are deterministic, i.e. there is no uncertainty in the determination of the vector of solutions and in the determination of the sensitivity analysis components. Results are limited to the inputs from which they were generated because the model was solved for one hypothetical herd at a fixed level of production, with a singular set of feed prices. The results are useful in the exploration of the uses of this optimization model in reducing the environmental impacts of livestock. For instance, when methane emissions were taxed at the level of current prices on the carbon credit market in the USA and Europe, diets were not significantly altered, and reductions of methane emissions were practically zero. These results suggest that current prices of the carbon dioxide equivalent are not sufficiently onerous to force a diet manipulation that would reduce methane emissions, i.e. force a change in the vector of solutions. In this context, the TAXM model can be used to investigate the effect of tax-based policies on diet formulation, demand for feeds by the dairy industry, environmental impacts and system profitability. In the scenario examined by Moraes *et al.* (2012), enforcing a tax-based policy, with current carbon credit prices used as tax values, did not alter diets and methane emissions; however, total costs were increased due to methane tax liabilities.

In the second model (REDM), methane emissions were reduced through a model constraint, which forced predetermined reductions in emissions. When emissions were reduced by 5%, 10% and 13.5% from the baseline scenario, dietary costs were greatly increased and nitrogen and mineral excretion were altered due to the selection of different feeds in diet formulation. When emissions were reduced by 13.5% from the baseline scenario, which was the maximum reduction for a feasible solution, nitrogen and potassium excretions were increased by 16.5% and 16.7%, and dietary costs were increased by 48.5%. The animal categories, which comprised the hypothetical dairy herd, were differently affected by the policy implementation. For example, when methane emissions were reduced by 10%, the mid- to late lactation cows group was the category that exhibited the largest proportional increase in nitrogen intake and when emissions were reduced by 13.5%, the dry cows group

exhibited the largest proportional increase in nitrogen intake. The identification of the animal categories, within a herd, that are more sensitive to environmental policies and the ones for which mitigation strategies can be implemented in a less costly manner was possible because model decision variables were indexed by animal category. Nutrient composition of diets was substantially altered when methane emissions were restricted. The neutral detergent fibre, crude protein and metabolizable energy dietary contents for the seven animal categories are presented in Fig. 6.1. Dietary fibre was substantially reduced and dietary protein and energetic density were increased when methane emissions were restricted. The reduction of methane emissions was achieved by the reduction in dry matter intake and by the reduction of the proportion of fibre in the diet, consequently forcing the model to select feeds with higher energy, protein and mineral contents. Mineral and nitrogen intake and excretion were altered with reductions in methane emissions, suggesting that, for our hypothetical herd, reducing methane emissions through dietary manipulation, especially through the reduction of dietary fibre, can alter and possibly increase nitrogen and mineral intake and excretion. Shadow prices were generated in the methane restriction constraint and used to represent the marginal costs of mitigating methane emissions. When plotted against the respective reductions in methane emissions (Fig. 6.2), shadow prices increased monotonically, suggesting that the cost of reducing the unit of methane emissions increases with larger reductions in methane emissions. The shadow prices generated by the model structure can be used in a cap and trade policy scheme, in which total methane emissions are capped but participants are allowed to trade residual emissions. The shadow price is the inputted cost attributed to the residual methane emission when a reduction is enforced; therefore, it can be used to input a trading value for those residual emissions. Emissions could then be traded between policy participants, for example in a carbon credit market. The marginal costs of reducing methane emissions were extremely high in all scenarios, which were consistent with the results from TAXM, where carbon market current prices did not reduce methane emissions through taxation.

This modelling framework is very flexible; the feed composition matrix, animal nutrient requirements, feed prices, environmental impacts and herd structure are easily changed to accommodate different production systems. The deterministic nature of the model and the limited data for which the model was solved limit the extrapolation of these results to the larger condition of dairies. However, this limitation is related to model inputs, and the model structure can be easily incorporated into whole-farm models, which can simulate different production systems with different input levels. Moreover, a more robust analysis can be performed using this modelling structure, with the incorporation of uncertainty in model inputs without compromising model linearity, for example through a Monte Carlo simulation study.

Final Considerations

Substantial progress has been made in the development of diet optimization techniques in recent years. Several extensions of the linear programming diet formulation model have been developed and implemented to optimize various diet formulation objectives. The incorporation of uncertainty in feed composition, feed costs and nutrient requirements have extended the deterministic approach of diet optimization. The development of multi-criteria programming models extended the application range of mathematical programming, enabling the derivation of efficient solutions for multiple objective functions. Concomitantly, a better understanding of feed nutrient availability has been achieved through the development of feeding systems in which nutrients are divided according to feed digestibility and animal production levels. Precision feeding strategies have brought a new perspective to the feeding of animals at the requirement level. The incorporation of animal variability in the determination of nutrient requirements has extended the concept of formulating diets for the average animal. Moreover, several mathematical and statistical models have been developed to determine environmental impacts of livestock production. In this context, the incorporation of animal characteristics, feed information and environmental impacts in a diet optimization model enables the joint minimization of diet costs and the environmental impacts from livestock production. These models

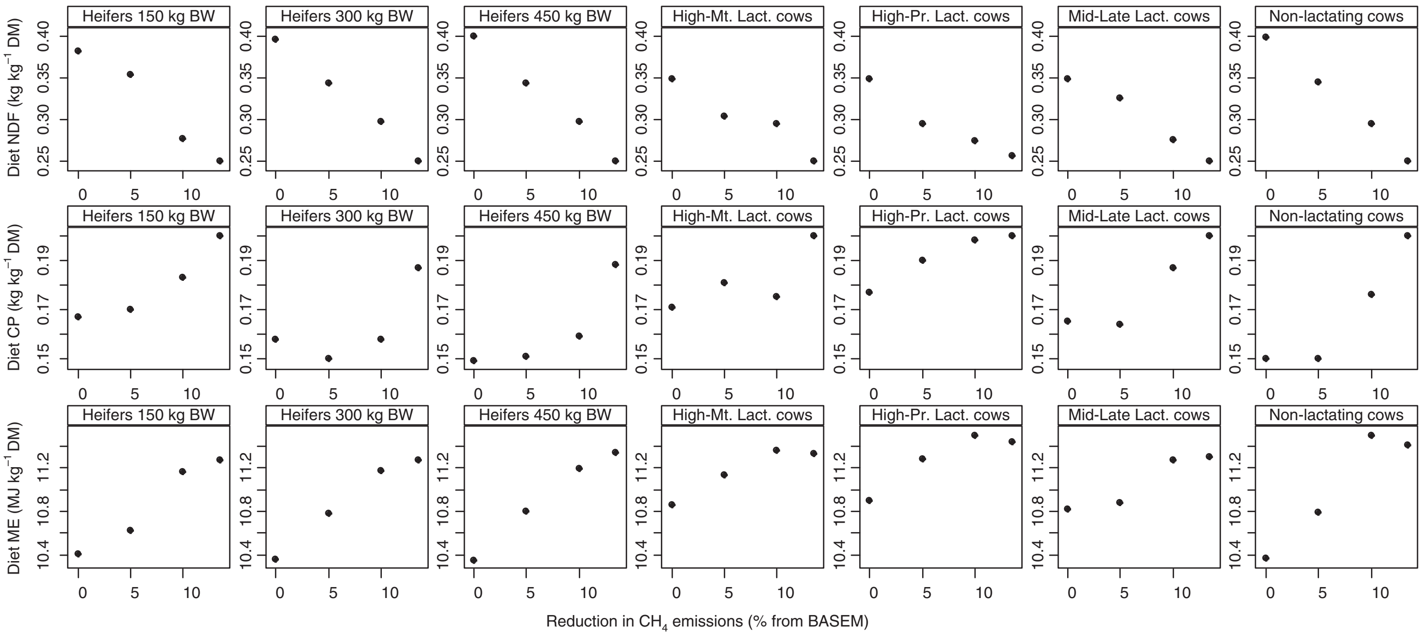

Fig. 6.1. Neutral detergent fibre proportion (kg NDF per kg DM), energetic density (MJ of ME per kg DM) and crude protein proportion (kg CP per kg DM) of diets formulated in the model with predetermined methane emission reductions (REDM) versus respective reductions in methane emissions from the baseline scenario. Note: ‘High-Mt. Lact. Cows’ represents the high-producing multiparous lactating cow group, ‘High-Pr. Lact. cows’ represents the high-producing primiparous lactating cow group and ‘Mid-Late Lact. cows’ represents the mid- and late lactation cow group. BASEM represents the baseline scenario. Other individual titles are self-explanatory. From Moraes *et al.* (2012).

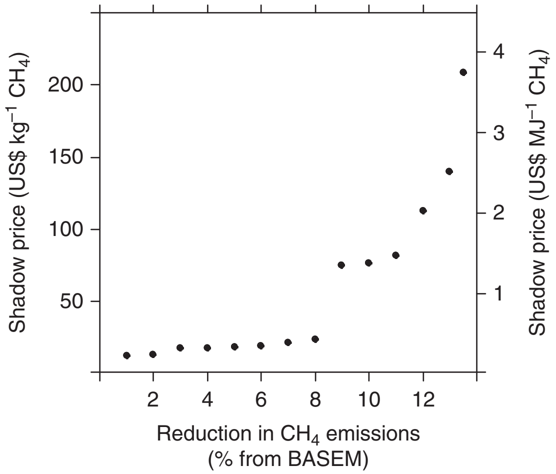

Fig. 6.2. Shadow prices of the methane restriction constraint of the model with predetermined emission reductions (REDM) versus respective reductions in methane emissions from the baseline scenario (13.5% was the maximum reduction in methane emissions for a feasible model solution). Note: BASEM represents the baseline scenario. From Moraes *et al.* (2012).

can be used by producers and policy makers in examining the effects of regulatory policies on the production system and in identifying strategies that mitigate environmental impacts in an optimal manner. As a result, management decisions, which comply with sustainable practices, can be identified and production systems in which minimal environmental impacts are generated can be established.

Summary

The livestock industry has received increasing pressure from society to deliver animal products at minimal environmental cost. Methane emissions from livestock, especially from ruminants, are listed as a substantial agricultural source of GHG emissions. Mineral and nitrogen excretions from livestock are now regulated in several countries. The livestock industry will be required to adapt to new regulations and shape its production system to meet the demands set by regulatory policies. The decision maker, either at the planning or at the operational level, will be required to make decisions that optimize animal production at minimal environmental cost. New standards will be set by consumers, increasing the demand for products from sustainable production systems and, thereby, increasing the cost of generating environmental impacts. For compliance with new standards, decision making models must incorporate the effects of environmental policies on producer utility, system sustainability and profitability. Feeding costs play a key role in profitability because they represent a substantial proportion of the total cost in a livestock production system. The selection of feeds, the purchasing of feeds and the formulation of diets are important activities involved in the feeding of livestock. Elements affecting any of these factors, such as feed prices, milk demand and regulatory policies, should be considered in feed management. Diet optimization models that incorporate environmental information can be used in the selection of the feed mix that minimizes environmental impact at minimal cost. In this study we present a variety of diet optimization modelling techniques and, most importantly, their applications addressing the environmental impacts in an optimization scenario. An example of the formulation of diets for a dairy herd, when environmental policies are enforced, is presented and the results are discussed to provide an application of model solutions and sensitivity analysis in examining mitigation strategies. In the scenario examined, enforcing a reduction of methane emissions from dairy cattle greatly increased diet costs, and mineral and nitrogen excretion were substantially altered.

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7 Sustainable Manure Management

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Introduction

Manure management has risen to the fore of livestock production concerns in the developing world, with implications to farming system design, function and profitability, as well as to environmental quality and human health. Sustainable manure management, i.e. manure management that balances the production, economic and environmental concerns of manure generation, handling, processing and end use, requires concerted investment of resources and time as well as a strategic approach to livestock production that often extends well beyond the farm gate. This chapter seeks to elucidate the concerns underlying manure management and the considerations of sustainable manure management. While we provide a broad review of salient issues, particular emphasis is placed upon emerging areas of concern and innovation in manure management.

To appreciate the challenges of manure management in modern livestock operations, it is instructive to consider the evolution of livestock production from traditional, diversified systems with a high degree of integration between feed and livestock production, to modern, specialized systems in which feed and livestock production are sometimes completely separate. The advent of commercial fertilizers largely replaced the role of livestock manure as a primary source of nutrients for crop production. Networks for transporting feed and forages have enabled the specialization and intensification of animal production, with the number of animals used for production of meat, milk and eggs increasing from ~10 billion in 1930 to 68 billion in 2002 (FAOSTAT, 2012). By 2000, it was estimated that livestock were generating 101 and 17 Tg of manure nitrogen (N) and phosphorus (P), respectively, per year (Bouwman *et al.*, 2009). While the quantity of manure nutrients generated over this period increased, the growing disintegration of livestock and crop production has left most modern livestock producers with a surplus of on-farm nutrients.

Even greater concentrations of manure nutrients are apparent at regional scales (e.g. Maguire *et al.*, 2005). Potter *et al.* (2010) determined the geographic distribution of manure N and P produced from the main livestock groups (cattle, buffalo, goats, sheep, pigs and poultry) and reported that the highest rates of N and P in manures produced are found in the USA, parts of South America, western Europe, East Africa, northern India, eastern China and New Zealand. Regions with concentrated animal production that also use fertilizers in crop production have

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high levels of excess N, P and potassium (K). In Europe, de Walle and Sevenster (1988) determined N balances for 11 EU countries and reported that all countries had a surplus of N (22–320 kg ha^{-1}) with five having a surplus greater than 100 kg ha^{-1}. A national nutrient balance for Korea, in 1995, showed that excess nutrients were in the order of 331, 386 and 406 thousand kg for N, P and K, respectively (Richard and Choi, 1999). In the USA, the number of confined livestock farms declined 50% from 1982 to 1997, while the number of confined animal units increased 10% (Gollehon *et al.*, 2001). This shift in production led to more excess on-farm nutrients (734,000 t N and 462,000 t P in 1997) and the separation of land from livestock. Approximately 20% of farm-level excess N and 23% of farm excess P exceeded the land assimilative capacity at the county level. To solve these nutrient distribution problems, substantial amounts of manure may need to be transported off farm and in some cases out of the region in which they are produced.

For many livestock producers, manure has become one of the most problematic areas of management under a growing litany of off-farm concerns. Concerns over long-term self-sufficiency related to inefficient recycling of manure nutrients are mounting. More acute are nuisance, environmental and health concerns associated with environmental emissions and runoff. Both large and small operations face these issues. Current major strategies for sustainable manure management include: (i) on-farm; and (ii) off-farm recycling of nutrients for crop production; (iii) recycling of nutrients as an animal feed ingredient; and (iv) export to non-agricultural uses (energy production, fibreboard, plastics).

Sustainable manure management implies more than improved use of manure nutrients in crop production, ranging from the effective use of the energy potential associated with manures, to the minimization of off-site transport of potential contaminants. How manure is handled, stored and land-applied affect its quality, including nutrient value and potential to pollute the environment. Manure treatment can also play a central role in changing manure quality to enhance its beneficial properties and minimize its potentially adverse properties. Therefore choosing the appropriate manure management system is essential in order to continue to have large-scale animal production that balances food production priorities with negative environmental and social consequences.

On-farm Manure Management Systems

Manure collection and handling systems enable livestock producers to utilize all the components in their manure management system efficiently. Unconfined livestock operations, i.e. grazing operations where dung is excreted on pastures or rangeland, are not included in this discussion. A typical manure management system will include some or all of the following components: (i) area where manure is produced (i.e. feedlot/dry lots, barns, other confinement buildings); (ii) manure treatment area including recycling of useable manure by-products (i.e. solids separator, digester, composting); (iii) manure transport (i.e. transfer of manure from collection to storage or treatments areas); and (iv) manure storage facility (i.e. manure tank, holding pond, stackhouse or other storage area). The purpose of manure collection and handling systems is to gather and move manure among the components of a manure management system efficiently and safely. This system can also incorporate technologies to fractionate the manure to improve its utilization and derive more value from it.

The type of equipment and procedures used to collect and handle manure depends primarily on the solids content of the manure, with the quantity of solids in manure varying with species and production system. Classification of manures on the basis of solids content varies, but most conventional definitions classify 'solid manures' as those with greater than 20% solids, including 'litters', which are predominantly generated by poultry operations and tend to contain greater than 70% solids, 'semi-solid manures' that contain 12–20% solids, 'slurries' that contain 4–12% solids and 'liquid manures' that contain less than 4% solids. The solids content of excreted manure is often changed by processes such as adding bedding, drying manure on a lot surface, adding washwater or dewatering the manure by solids separation.

Solid and semi-solid manures are usually collected using scrapers, box scrapers, blades, front-end or skid-steer loaders, or similar devices.

These manures are typically transported by truck and directly land applied, stacked for storage, or composted. Slurry manure is typically generated where little bedding is added to the excreted manure/urine. The simplest manure collection arrangement is slotted or perforated flooring over a collection tank. Slurry manure can also be collected using scrapers or vacuums. Slurry is usually pumped and stored or treated, in some cases it is directly land applied. Liquid manures generally result from the addition of washwater or rainwater to manure. Flush systems are common where manure is flushed from the confinement building using either fresh or recycled water. Runoff from lot surfaces can be treated and stored in holding ponds. In most cases, the liquid is blended with clean water and used as irrigation water.

The characteristics of the manures collected in these systems will vary with species and production system and determine the nutrient value of the manure, the potential for gaseous emissions, the potential for energy generation, and the extent of treatment/processing needed to transform the manure into value added products. Tables 7.1 and 7.2 provide some manure characteristics as excreted by livestock and poultry, while Table 7.3 provides 'as removed' manure characteristics for the main livestock and poultry groups. It is important to remember that these values can change greatly based on animal feeding (see Chapter 6, this volume) and manure handling and storage. As demonstrated in these tables, manures contain valuable nutrients that can be recycled in the crop–animal system and solids that can be converted to energy and other valuable manure by-products.

Land Application of Manures for Crop Production

Land application of manure to support on- and off-farm crop production is common worldwide and a fundamental attribute of sustainable manure management systems. The positive contributions of manure to soil fertility and soil tilth are well established (e.g. Williams and Cook, 1961; Bationo and Mokwunye, 1991). Crop response to recent application of manure is generally positive, with yields stimulated by macro- and micronutrients in manure. Over the longer term, manure can substantially augment soil organic matter and soil structural properties that stabilize aggregates, increase water-holding capacity and improve rainfall infiltration. Despite these benefits, land application of manure can result in adverse impacts, many of little immediate concern to crop production, which complicate land application and must be considered in devising sustainable manure management strategies. Westerman and Bicudo (2005) listed eight challenges for integration of manures into agricultural production, which included:

- Regional imbalances of nutrients (e.g. not enough land on the farm to apply nutrients produced by animals on the farm);
- Imbalances of nutrients in manure compared with crop needs;
- The relatively low nutrient concentration compared with chemical fertilizers;
- The variability in nutrient content, difficulty in quickly determining nutrient content, and predicting the availability of nutrients to growing crops;
- The often bulky nature of manures making it more difficult to haul and spread consistently;
- Possible transfer of weed seed;
- Satisfying environmental regulations on application amounts, application timing, and application methods; and
- Possible environmental concerns, such as emission of ammonia (NH_3) and greenhouse gases (GHG), odour, pathogens and pharmaceutically active compounds (PAC).

Given these concerns and the growing regulatory and paperwork burden of land-applying manure in many areas of the developed world, manure's value can readily turn from an agronomic resource to a perceived liability (Kleinman *et al*., 2012). Manure's bulky nature, heterogeneous nutrient forms, concentrations and ratios and adverse qualities (e.g. weed seed, odour, pathogens) make manures imperfect fertilizer substitutes. As a result, pound for pound, manure nutrients are more costly to transport than commercial fertilizers, often making the economic value of manure greatest near the point of generation, i.e. the livestock barn.

Land application of manure for crop production involves an array of potential environmental and human health concerns, the latter of which are discussed in greater detail below. Sustainable

Table 7.1. Estimated typical manure characteristics as excreted by meat producing livestock and poultry. (Source: ASABE Standard D384.2.)

Animal type and production grouping	Total manure (kg per finished animal)	Moisture (% w.b.)	Total solids	Volatile solids	COD	BOD	N	P	K	Assumed finishing time period (days)
			(kg per finished animal)							
Beef										
Finishing cattle	4500	92	360	290	300	67	25	3.3	17.1	153
Swine										
Nursery pig (12.5 kg)	48	90	4.8	4.0	4.4	1.5	0.41	0.068	0.16	36
Grow-finish (70 kg)	560	90	56	45	47	17	4.7	0.76	2.0	120
Poultry										
Broiler	4.9	74	1.3	0.95	1.05	0.30	0.053	0.016	0.031	48
Turkey (male)	36	74	9.2	7.4	8.5	2.4	0.55	0.16	0.26	133
Turkey (females)	17	74	4.4	3.5	4.0	1.1	0.26	0.074	0.11	105
Duck	6.5	74	1.7	1.0	1.4	0.28	0.062	0.022	0.031	39

COD, chemical oxygen demand; BOD, biochemical oxygen demand.

Table 7.2. Estimated typical manure characteristics as excreted by all other livestock and poultry. (Source: ASABE Standard D384.2.)

Animal type and production grouping	Total manure (kg per day per animal)	Moisture (% w.b.)	Total solids	Volatile solids	COD	BOD	N	P	K
			(kg per day per animal)						
Beef									
Cow (confinement)	–	88	6.6	5.9	6.2	1.4	0.19	0.044	0.14
Growing calf (confinement)	22	88	2.7	2.3	2.3	0.52	0.13	0.025	0.085
Dairy									
Lactating cow	68	87	8.9	7.5	8.1	1.30	0.45	0.078	0.103
Dry cow	38	87	4.9	4.2	4.4	0.626	0.23	0.03	0.148
Heifer – 440 kg	22	83	3.7	3.2	3.4	0.54	0.12	0.020	–
Swine									
Gestating sow – 200 kg	5.0	90	0.50	0.45	0.47	0.17	0.032	0.009	0.022
Lactating sow – 192 kg	12	90	1.2	1.0	1.1	0.38	0.085	0.025	0.053
Boar – 200 kg	3.8	90	0.38	0.34	0.27	0.13	0.028	0.0097	0.0176
Poultry									
Layer	0.088	75	0.022	0.016	0.018	0.005	0.0016	0.00048	0.00058

COD, chemical oxygen demand; BOD, biochemical oxygen demand.

Table 7.3. As removed characteristics by all other livestock and poultry. (Source: ASABE Standard D384.2.)

	Moisture	TS	VS	TKN	TAN	P	K	Ca	Na	Mg	S	Zn	Mn	Cu
	(% w.b.)		(%TS)	(% w.b.)					(ppm w.b.)					
Beef														
Earthen lot	33.1	67.2	30.2	1.180	0.10	0.50	1.25	1.21	3012	3650	2841	85	393	14
Dairy														
Scraped earthen lot	54	46	–	0.70	–	0.25	0.67	0.45	311	100	–	1.2	1.4	0.2
Scraped concrete lot	72	25	–	0.53	–	0.13	0.40	0.31	32	9	–	0.4	0.7	0.1
Lagoon effluent	98	2	52	0.073	0.08	0.016	0.11	0.04	7	3	–	0.9	1.4	2
Swine														
Finisher – slurry wet dry feeders	91	9	–	0.7	0.5	0.21	0.24	0.25	380	–	400	85	–	50
Slurry storage dry feeders	93.9	6.1	–	0.47	0.34	0.18	0.24	0.25	380	–	180	68	–	30
Flush building	98	2	–	0.2	0.14	0.07	0.17	0.04	300	290	155	33.6	14.4	31.2
Agitated solids & water	97.8	2.2	–	0.10	0.05	0.06	0.06	0.08	215	300	180	44.4	15.6	19.2
Lagoon surface water	99.6	0.40	–	0.06	0.04	0.02	0.07	0.01	215	55	37	3.6	1.2	2.4
Lagoon sludge	90	10	–	0.26	0.07	0.25	0.07	0.04	191	132	79	22	80	90
Poultry														
Broiler litter	31	70	70	3.37	0.75	0.60	1.37	1.82	–	–	–	–	–	–
Turkey litter	30	–	–	2.18	–	0.33	1.23	5.0	–	–	–	–	–	–

TS, total solids; VS, volatile solids; TKN, total Kjeldahl nitrogen; TAN, total ammoniacal N.

management of manures requires concerted effort to avoid the unintentional, adverse consequences of a seemingly prudent agronomic practice. In many areas of the developed world, land application of manure is regulated. Regulations range from international directives aimed at improving environmental quality (e.g. EU Nitrates Directive; European Commission, 2010) to local rules aimed at preventing nuisance complaints (e.g. municipal odour ordinances; SRF Consulting, 2004). Site selection and manure application method, rate and timing are the primary factors controlling these adverse impacts (Fig. 7.1).

Site selection

Site selection represents the first step in decision making when land-applying manures. The potential for nuisance concerns is often a primary site selection factor. In general, nuisance concerns (e.g. odour, flies) can be readily addressed by simply avoiding sites that are in proximity to or upwind of potential sources of offence (e.g. housing tracts). Site selection for manure application must weigh potential trade-offs (e.g. sites that conserve manure N may be prone to P loss in surface runoff) and reflect farming system concerns (e.g. field availability, crop requirement, manure storage and handling capabilities) for a particular physiographic context (site selection in sloping upland landscapes will feature different priorities than site selection in flat coastal plain landscapes). An array of nutrient management decision support tools have been developed over the past several decades and are now in widespread use across the USA, Europe and South Pacific. These tools generally consider local climate, site hydrology (e.g. seasonal high water table), soil properties (e.g. erodibility, leaching potential), field management (e.g. tillage system) and delivery factors (e.g. field buffers) as indicators of the potential

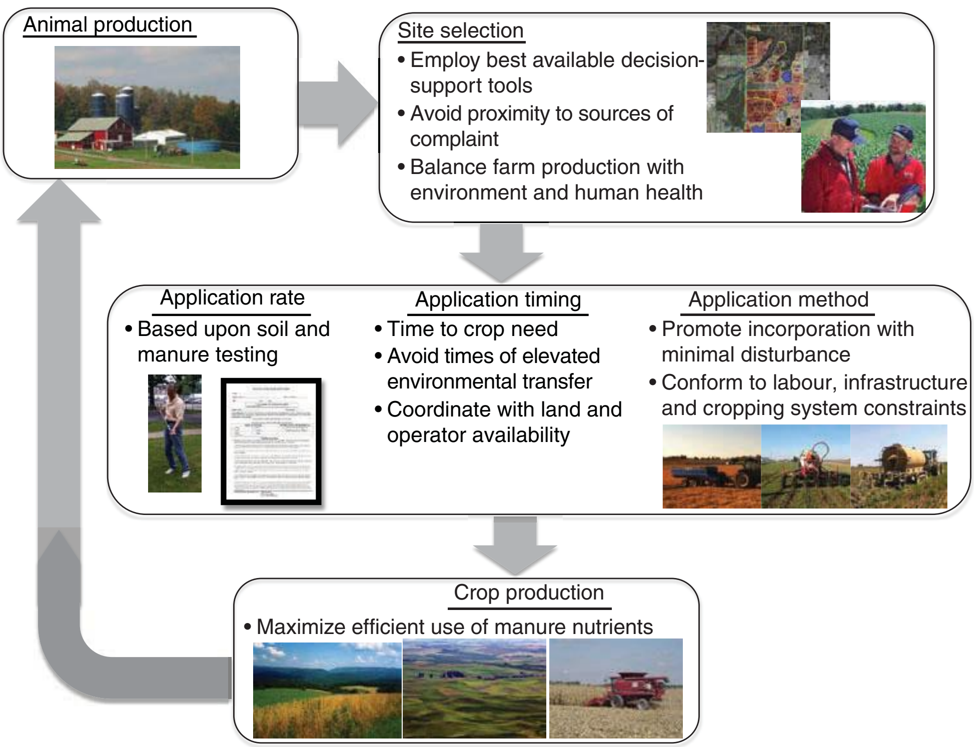

Fig. 7.1. Linking manure and crop production through the central components of a successful land application programme.

off-site transport of manure nutrients (Sharpley *et al.*, 2003; Delgado and Follett, 2011). Such tools offer strategic decision support for land application of manure. More recently, advances in the weather forecasting models have been used to develop prototype tools that provide daily decision support on whether sites are suitable to receive manure based upon pollution potential (Dahlke *et al.*, 2008; Melkonian *et al.*, 2008; Buda *et al.*, 2013).

Manure application methods

The fate of manure constituents and the impact of land application on crop, soil, air and water quality can be profoundly affected by manure application methods. Broadcast application of manure is the most ubiquitous application method of both liquid and dry manures alike. Much of the reported inefficiency in delivering manure nutrients to crops stems from the prevalence of broadcast application. Generally, broadcasting manures exacerbates emissions of NH_3 (Dell *et al.*, 2011; Pfluke *et al.*, 2011) and odour (Brandt *et al.*, 2011) and losses of P in runoff (Johnson *et al.*, 2011), and promotes severe vertical stratification of nutrients in soil. Incorporation of manure into soil at time of application generally improves nutrient use relative to broadcast application. However, incorporation has historically been accomplished by tillage, conflicting with no-till and perennial forage management systems. In recent years, an array of low disturbance technologies has emerged from Europe and the USA that place manure below the canopy of a perennial crop or into the subsoil with minimal disturbance to surface cover. Low disturbance technologies range from those that band manure below the canopy of perennial forages (trailing hose, trailing shoe) to those that inject manure into the soil (chisel injection, disc/coulter injection, high pressure injection) to those that lightly till or perforate the soil to improve infiltration of applied manure into the surface (aeration, low-disturbance vertical/strip tillage). Many configurations of these technologies exist, such that there are considerable options to adapt low-disturbance applicators to handle the specific requirements of existing machinery/manure handling systems, cropping systems and soil conditions. Barriers to the adoption of these specialized technologies often revolve around the capacity and speed of application relative to broadcast application and perceived cost. However, Rotz *et al.* (2011) demonstrated that the cost of adopting some low disturbance applicators on small dairy farms in the north-eastern USA was either neutral to slightly profitable in comparison with traditional broadcast methods. A comprehensive review of low-disturbance manure application technologies and their associated impacts on surface residue, nutrient loss, erosion, odour and crop response is offered by Maguire *et al.* (2011).

Poor accounting for manure nutrients often results in excess application of manures for crop production. *Manure application rate* is strongly tied to fugitive losses of nutrients to air and water over the short term (Thompson *et al.*, 1990; Kleinman and Sharpley, 2003). Over the long-term, repeated applications of manure in excess of crop requirement can overwhelm the soil's natural buffering capacity, resulting in chronic contributions that cannot be readily controlled with manure management options (Kleinman *et al.*, 2011). Sustainable manure management must consider realistic yield expectations and credit existing sources of nutrients (past manure application, recent legume crops). Testing of manure and soil are necessary to optimize manure application rates, although use of book values may represent a significant improvement over the status quo in many instances. Application rates should be adjusted to account for the significant differences in manure nutrient use efficiency of different application methods. For instance, immediate incorporation of manure may conserve more than 60% of the plant available N in manure that would be lost with broadcast application (Beegle, 2012).

Timing of manure application

Timing of manure application represents one of the most difficult logistical aspects of land application programmes for manure recycling. Application of manure shortly before crop planting is recommended to ensure that manure nutrients are employed by the growing crop. Long periods between manure application and crop growth lower the nutrient use efficiency as

manure nutrients are removed by environmental processes. Avoiding manure application during periods of high potential for runoff or drainage is a key aspect of preventing losses of manure constituents to water (Walter *et al*., 2001). Timing can also be used to optimize nutrient delivery; broadcast application immediately prior to a light rain can significantly translocate manure nutrients into the soil, providing many of the benefits of the incorporation techniques described above. However, a variety of factors prompt farmers to apply at other times, and these factors must be considered if manure application timing is to be improved. The absence of sufficient manure storage is most often cited as a cause of poor timing. Older, smaller confinement facilities sometimes lack the infrastructure to store manure temporarily and continue to rely upon daily application of manure, year round (Dou *et al*., 2001). Even when adequate storage exists, severe weather conditions (e.g. extreme precipitation associated with hurricanes) may overwhelm open storage structures and drive farmers to land-apply manure when conditions are poor. Alternatively, site access may push farmers to land-apply during periods when crops have been recently harvested, soils are trafficable (waterlogged soils are frozen), or no other fields are available to receive manure. Ultimately, farming education, expanded options for land application of manure (including off farm export), well integrated farming systems and accurate, short-term decision support (e.g. the forecasting tools described above for site selection) are all key to ensuring prudent timing of manure application for crop production.

Manure as a Livestock Feed Ingredient

As manures contain valuable nutrients and trace minerals, one option for utilization of manures (normally collected in dry systems) is the recycling of nutrients as an animal feed and as a nutrient source in aquaculture. Smith and Wheeler (1979) reported that animal excreta products contain 48% to 73% total digestible nutrients and 20% to 31% crude protein and therefore the nutrient content of manure has been shown to be three to ten times more valuable as animal feed than as plant nutrients. Utilizing animal manures as feed nutrients has many benefits including decreases in potential pollution and feed costs and better utilization of essential mineral sources. Ruminants are particularly ideal for the feeding of manures due to rumen microbiology and their ability to utilize fibre, non-protein N and nucleic acids. The most valuable manure for protein supplement in feedstock is poultry litter, due to the high concentration of nutrients. When processed by an acceptable method, poultry litter is an economical and safe source of protein, minerals and energy for beef cattle and swine (Carter and Poore, 1995; Akinfala and Komolafe, 2011). The most common methods for processing animal manures for producing feed are: drying, composting, ensiling, deep stacking, chemical treatment and extrusion-pelletizing (Arndt *et al*., 1979; Carter and Poore, 1995). If used as a feedstock, manures must be collected frequently to reduce losses of valuable N, as NH_3 volatilization from manures and litters happens quickly after excretion. In addition, the manure must be treated (composted, ensiled, chemical or heat treatment) to destroy pathogens and reduce odours to improve animal acceptability. It is also important to obtain accurate nutrient composition of manures if used as an animal feed as there is great variability in manure nutrient contents and they may differ significantly from published values (Zinn *et al*., 1996).

Manure can also be used in aquaculture systems, not directly as a feed, but as a fertilizer to enhance algae and other aquatic plant growth, which then serves as a feedstock. In China, animal manures have traditionally been used as fertilizer for fishponds and integrated fish farming and livestock production is common (Edwards, 1980). One of the more common systems is integrated poultry–fish farming which combines poultry production with fish culture where the spilled feed and manure from the poultry system are inputs into the fish subsystem (Sinha, 1985). The recycling of nutrients in the system allows for intensification of production and reduction of the environmental impact of production (Costa-Pierce, 2002).

Despite compelling reasons for recycling manure as a feedstock in livestock and fish production, food safety (e.g. bovine spongiform encephalopathy) and animal welfare concerns

are the basis for campaigns, and laws (Canada, EU), targeting the feeding of processed manures to livestock. Ultimately, public acceptance of this strategy will determine its contribution to sustainable manure management.

Manure Treatment Technologies and Non-agricultural Uses

In order to address some of the challenges related to use of manures for agricultural production and, in some cases, to produce more value added products from manures, there is a variety of manure treatment technologies available for on-farm use. Many of these manure treatment technologies have been used in one form or another for many years, while others are recent solutions. Often, technologies are used in combination to create a system that can be tailored to the manure management plan of the farm, climate or other factors (i.e. potential alternative end uses or by-product generation). Therefore, treatment technologies are generally selected to meet specific treatment goals on the farm. These treatment goals may include nutrient reduction or capture (primarily N and P), emissions reduction (including GHG, bioaerosols, NH_3, odours), volume reduction, energy recovery, pathogen reduction, and adding value to the manure. Manure treatment technologies are often linked together to address several treatment goals and challenges faced by animal producers such as excessive nutrients on farm, manure runoff and odour. Burton and Turner (2003) provide an excellent detailed discussion of many of these practices. Figure 7.2 presents a flow diagram of potential treatments and products that can be utilized in and derived from a manure management system. As some of the treatments/products are specific to solid versus liquid systems, they will be discussed separately below.

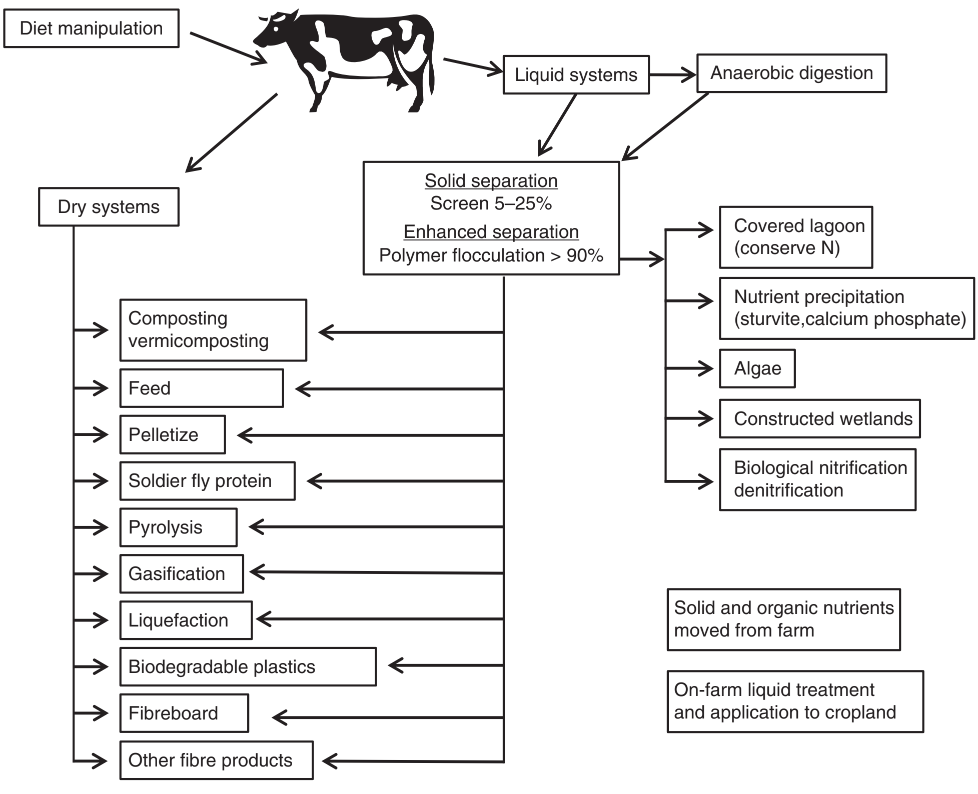

Fig. 7.2. Manure management treatments and technologies. Adapted from Szögi and Vanotti (2003).

Solid manure systems

Depending on the type of livestock raised and the production system, a large percentage of on-farm manure may be handled as a solid. Production systems that produce mainly solid manures are broiler and turkey operations, beef feedlots and dry-lot dairies. In addition, solid manures can enter the system via solid separation of slurries and liquid manures. As the moisture content in these manures tends to be low they are good substrates for composting, pelleting and for use in thermochemical conversion.

The need to move nutrients off farm has resulted in much interest in *composting* manures to reduce bulk, concentrate nutrients, reduce odour, kill pathogens and weed seeds, and have a stabilized product for transport (Westerman and Bicudo, 2005). In addition, composting manure has also been shown to degrade antibiotics effectively thereby reducing the potential for transport following land application (Kim *et al.*, 2012; Selvam *et al.*, 2012). The composted material is more uniform and easier to handle than raw manure, providing a source of slow release nutrients and therefore has commonly been used for years on many production facilities. There are several methods for composting manures: passive composting, aerated composting, windrow composting, in-vessel composting and vermicomposting. Passive composting is probably the most common method used today because it involves simply stacking manure (and other feedstock) and leaving them to compost over a long period. Very little, if any, activity is performed on the pile once it has been constructed. Aerated static pile composting modifies the passive composting technique by using blowers to supply air to the composting manure. This process does not involve turning and/or agitation of the piles. Electronic feedback controls are often used to monitor the pile temperature and control the operation of aerating blowers. Windrow composting is similar to passive composting although the piles of manure are turned or aerated by mechanical equipment to maintain optimum conditions. Manures are placed in long rows and are mechanically turned at frequent intervals in the composting process. In-vessel composting refers to any type of composting that takes place inside a structure, container or vessel. Each type of vessel system relies upon mechanical aeration and turning to enhance and decrease the duration of the composting process. All of these systems require a greater investment in manure management as manure/compost is moved several times and needs to be mechanically turned or aerated in some way; in some cases these costs can be prohibitive for on-farm adoption.

One of the major disadvantages of composting raw manures is the loss of N as NH_3 – a valuable nutrient for crop production and an air quality concern, as well as a loss of carbon (C), as carbon dioxide (CO_2) and methane (CH_4), which is a valuable soil conditioner. During the thermophilic phase (high temperature) of composting, much of the manure N is lost mainly as NH_3. Nitrogen losses can range from 3% to 60% of total initial N (Bernal *et al.*, 2009) with the majority of the N lost during the first 4 days of composting (Jiang *et al.*, 2011). The loss of organic matter or C has been shown to range from 9–81% of initial OM depending on the manure type and bulking agent used (Bernal *et al.*, 2009) with the greatest loss of C occurring later in the composting process (Jiang *et al.*, 2011). Countermeasures to reduce the loss of N during the composting process can improve the fertilizer value of composts, and several management options are available. The addition of a readily available C source such as molasses has been shown to reduce NH_3 losses as more N is stabilized in the microbial biomass (Liang *et al.*, 2006). Additives such as zeolite and biochar have been shown to reduce N losses by up to 52% (Steiner *et al.*, 2010; Luo *et al.*, 2011). Fukumoto *et al.* (2011) demonstrated that the use of struvite precipitation and nitration promotion in the composting process of swine manure reduced total N losses by 60%.

Vermicomposting is a process that relies on earthworms and microorganisms to help stabilize active organic materials and convert them to a valuable soil amendment and source of plant nutrients. As the process is mesophilic (moderate temperature) less N is lost during the process leaving a lower C:N ratio, which improves its value for agricultural uses (Lazcano *et al.*, 2008). Due to the lower composting temperature, manures do not undergo thermal stabilization that eliminates pathogens. Therefore, one potential drawback to the use of vermicomposting for treating animal manures is the presence

of human pathogens, which could restrict the use of vermicompost as an organic fertilizer (Aira *et al*., 2011). To circumvent this problem, thermophilic composting as a pretreatment to vermicomposting is also being used to reduce pathogens. Mupondi *et al*. (2011) reported that a pre-composting period of 1 week was found to be ideal for the effective vermicomposting of dairy manure.

In order to increase the bulk density, nutrient density and particle size uniformity of manures there has been interest in *pelletization*. By pelletizing manures, a more nutrient-dense product is available for transport, thereby enabling a larger land area to be utilized for land application (Hammac *et al*., 2007). Pelletization of manures can be done with dry manures such as poultry litter or liquid manures with the addition of dry substances (Heinze, 1989). In 2001, the world's largest pelletization plant, Perdue-AgriRecycle Poultry Manure Pelletization Plant, was opened on the Delmarva Peninsula to process 95,000 t of manure a year. This was a joint effort between Perdue, one of the largest US poultry producers, and the State of Delaware to help address regional nutrient accumulation issues. The product produced in the pelletization process is shipped around the world for use as fertilizer and fish feed. In Ireland, technology was developed to blend composted biodegradable farm wastes such as pig manure, spent mushroom compost and poultry litter with dried blood or feather meal with mineral supplements, which was then pelletized to produce an organo-fertilizer with specific N:P:K target ratios that was pathogen free (Rao *et al*., 2007). These designer organo-fertilizers are one way to add additional nutrient value to manure and increase their marketability.

Solid manures can also be used for *thermochemical conversion* to produce biogas. Technologies that burn manure to produce energy or treat manure to produce fuels are classified as 'thermochemical conversion', and include direct combustion (burning with excess air to produce heat), pyrolysis (thermal treatment in the absence of air, resulting in the production of pyrolysis oil and a low-BTU gas), gasification (thermal treatment at higher temperatures in an oxygen-restricted environment to produce a low- to medium-BTU gas) and hydrothermal liquefaction (thermal conversion of solids in a liquid stream to oils and char for separation and use as a fuel). The fuels that are products of pyrolysis and gasification can be used in boilers and engines. By-products from pyrolysis, such as biochar, are also being investigated as a soil amendment for C storage. Cost estimates for these technologies vary widely and do not always include the costs of pretreatment, drying and fuel preparation, or post-treatment, gas clean-up, electrical generation, and emissions controls. In general, it appears the value of the heat and energy alone does not provide sufficient financial incentive for a thermochemical conversion facility. Additional income streams that might make the technology more economically appealing do not currently exist but could include a combination of tipping fees collected for accepting manure solids, renewable power production tax incentives and the recovery of value from the ash. Liquid manures can also be used in thermochemical conversion to produce energy via direct liquefaction, aqueous-phase gasification and combined pyrolysis/gasification (Cantrell *et al*., 2007).

Slurry and liquid manure systems

Livestock production facilities that house animals in confinement buildings typically generate large amounts of both slurry and liquid manure. In many cases, slurry and liquid manure undergoes some form of *solid–liquid separation* (removal of organic and inorganic matter) prior to storage. Objectives for removing solids include removal of nutrients for transport off-site, removal of larger particles to make liquid transfer more efficient, and removal of organic material to reduce volatile emissions. Separation efficiency depends on the particle size distribution in the influent, the characteristics of the treatment technology and the treatment time. Separation devices can utilize gravity flow, have few moving parts and require little management effort, or they can utilize pumps and motors and require intensive management. Mechanical separators include: stationary inclined screen; vibrating screen; rotating flighted cylinder; rotating cone; piston; liquid cyclone; and roller, belt, screw or filter presses. Gravity separators include settling basins, ponds and weeping walls.

Chemical flocculants have also been used in conjunction with solid–liquid separation systems in order to improve separation efficiencies. For example, solid separation using screens has low efficiency with solids removal rates of 5 to 15% (Vanotti and Hunt, 1999), whereas solid separation using screens with a flocculant agent can remove >90% of total and volatile solids and >70% of chemical oxygen demand (COD) and total N and greater than 50% of TP (García *et al*., 2009; Paz Pérez-Sangrador *et al*., 2012). Precipitation or flocculation in a treatment cell where the material can be harvested is beneficial. Although there are many separation techniques available for use on livestock farms, there is still a need to improve the cost effectiveness of these technologies. Once manure has gone through a solid separator, the remaining liquid is either transferred to a storage pond where it is kept until land application, or it can be further processed to produce recycled cleaner water, energy and other by-products.

Once the liquid fraction is transferred to a storage pond, basin or lagoon, there can be additional losses of N due to NH_3 volatilization as well as the generation of CH_4 and volatile organic compounds (VOCs; responsible for odour) due to the development of anaerobic conditions. One way to minimize these emissions is to utilize a *cover*. Guarino *et al*. (2006) tested several permeable covering systems (maize stalks, wood chips, vegetable oil, expanded clay, wheat straw) to reduce emissions from livestock slurry tanks and lagoons. They reported reductions of NH_3 emissions from swine and dairy slurry in the range of 60–100% with 140-mm solid covers or 9-mm liquid covers. Miner *et al*. (2003) reported that a permeable polyethylene foam lagoon cover reduced NH_3 emissions by approximately 80% on an anaerobic swine lagoon. Floating an impermeable cover over the surface of a lagoon or pond can also capture up to 80% of methane and reduce odours. The trapped gas can be flared or used to produce heat or electricity. Craggs *et al*. (2008) reported that placing a floating polypropylene cover on anaerobic swine and dairy ponds yielded biogas recoveries of 0.84 m^{-3} m^{-2} day^{-1} and 0.032 m^{-3} m^{-2} day^{-1}, respectively. They estimated that this could produce 1650 and 135 kWh day^{-1} from fully covered anaerobic swine and dairy ponds. Permeable surface covers not only act as a physical barrier to gas transport, but can also support microbial communities that are capable of utilizing reduced gases emitting from the slurry (Petersen and Miller, 2006) providing additional mitigation benefits.

When land for application of liquid manures is limited, treatment systems that remove N via *biological nitrification/denitrification* have been employed. Uncovered anaerobic lagoons have commonly been used to treat livestock wastewaters with the main treatment focusing on N volatilization and reduction of solids. While the goal is to reduce N by conversion to N_2 gas there is still a large amount of NH_3 volatilized into the atmosphere. These lagoons are also large sources of CH_4 as the solids are broken down in anaerobic conditions, and they have been identified as one of the major GHG emitting sources in livestock production systems (Leytem *et al*., 2011). Systems that capture the CH_4 and use it for energy generation or flare the CH_4 help mitigate this impact (see below). The design of batch reactors to convert NH_4-N to N_2 has also been an area of research (Loughrin *et al*., 2009; Wang *et al*., 2010). Related to this is the anaerobic ammonium oxidation technology where ammonium (NH_4^+) is oxidized to N_2 and has been shown to remove up to 92% of NH_4-N from swine manure effluent (Molinuevo *et al*., 2009). Another cost effective and passive method for treating wastewaters is the use of constructed wetlands which are primarily designed to remove N prior to land application through plant uptake and denitrification. These constructed wetlands also have the added benefit of reducing the total suspended solids (TSS), COD and P, which are important from a water quality standpoint. A marsh-pond-marsh constructed wetland was shown to remove up to 51% of TSS, 50% of COD, 51% total N and 26% total P from swine wastewater (Poach *et al*., 2004). Constructed wetlands can also release N as NH_3, although this has been shown to be a relatively small portion of total N loss (Poach *et al*., 2002). The main limiting factor for denitrification in these systems is the conversion of N to nitrate (Hunt *et al*., 2006). While these management strategies can reduce the potential for water quality impairment from over-application of N, manure management strategies that are designed to waste valuable N are difficult to justify as replacing the lost N via chemical fertilization requires considerable expenditure of energy.

One of the more common energy production/capture systems for liquid manures is *anaerobic digestion*. Anaerobic digestion is a natural biological process by which bacteria break down organic matter in an oxygen-free environment with moisture content of 85% or higher. The process produces 'biogas', inorganic salts and residual organic material. The biogas consists of CH_4, CO_2 and trace amounts of other gases including hydrogen sulphide (H_2S). Biogas can be burned to produce heat or to power an electric generator. Zaks *et al.* (2011) estimated that anaerobic digesters have the potential to generate 5.5% of US electricity and mitigate 151 Mt CO_2e, mostly from CH_4 abatement. Rico *et al.* (2011) reported that anaerobic digestion of liquid dairy manure at 37°C produced 1.47 m^3 biogas (m^{-3} day^{-1}) and 1 m^3 CH_4 (m^{-3} day^{-1}) and could provide 2% of the total electrical power in the region of Cantabria, Spain. Marañón *et al.* (2011) reported that anaerobic digestion of cattle slurry on dairy farms in northern Spain could provide enough CH_4 to fulfil the farms' energy requirements and in some cases provide a surplus that could be used for heating and that annual GHG emissions savings ranged from 978 to 1776 kg carbon dioxide equivalents (CO_2e) per year due to reductions in CH_4 emissions during slurry storage. The amount of biogas produced and the percentage of residual organic matter depends on the duration of the anaerobic digestion process and factors such as temperature, moisture, nutrient content and pH. The residual organic material can be used for animal bedding, a soil amendment, or value added products such as fibreboards and other building materials. Additional benefits of anaerobic digestion are the breakdown of VOCs responsible for odour, and the destruction of weed seeds and pathogens. Digestion can occur in anaerobic lagoons or in engineered systems. The types of anaerobic digester technology available include: covered anaerobic lagoons, plug-flow digesters, completely-stirred tank reactor, upflow anaerobic sludge blanket and anaerobic sequencing batch reactor. Due to the large capital investment, initial set-up costs and expense of running a digester, they are not always economically feasible, particularly in areas with low energy prices. Yet, co-digestion of manure and other biomass is a potential way to improve the economics of digesters. The co-digestion of dairy manure and food processing wastes increased biogas production by 110% and tripled gross receipts on a commercial dairy (Frear *et al.*, 2011). The addition of vegetable waste in the anaerobic digestion of swine manure increased methane yield up to threefold (Molinuevo-Salces *et al.*, 2012).

Once manure has gone through a solid separation process or through a digester, the effluent can be treated to capture valuable nutrients in the liquid stream and concentrate them to generate a more valuable fertilizer source. A common technology to capture P, NH_4^+ and K is the use of *struvite* (magnesium ammonium phosphate) precipitation or P capture can be accomplished with hydroxylapatite (calcium phosphate) formation as well. Struvite precipitation has been found to remove 70–85% of P, 56–95% of NH_4^+ and <10% of K (Zeng and Li, 2006; Song *et al.*, 2011; Yilmazel and Demirer, 2011). However, a large amount of NH_4-N can be lost via volatilization (Song *et al.*, 2011). The use of hydrated limes to remove P from wastewaters has been shown to remove >90% of P (Vanotti *et al.*, 2003; Szögi and Vanotti, 2009).

The use of *algae* and other photo-bioreactors can also remove significant amounts of N and P, although these require light and the end-product may need significant processing before the nutrients can be re-used efficiently. The culturing of microalgae for biofuels production using wastewater as a nutrient source is also an area under investigation (Lam and Lee, 2012). Chen *et al.* (2012) demonstrated that non-filamentous green algae were able to tolerate high nutrient loads and could recover nutrients from wastewater from anaerobic digestion. It has been reported that up to 98% of N and 76% of P can be removed from wastewaters (Kebede-Westhead *et al.*, 2006; Chen *et al.*, 2012). Singh *et al.* (2011) reported a maximum biomass productivity of 76 mg l^{-1} day^{-1} for microalgae grown on poultry litter anaerobic digester effluent with a 60% and 80% removal rate of total N and P, respectively, from the effluent. The algae contained 39% protein, 22% carbohydrates and <10% lipids, making it a good animal feed supplement. The processed algae have also been tested as a slow release fertilizer (Mulbry *et al.*, 2005, 2007).

In addition to nutrient capture, other products can be obtained or made from solid separated

materials, liquid effluent, digester effluent and post-digester solids. *Fibreboard* and building materials can be manufactured utilizing digested solids in place of sawdust (Winandy and Caia, 2008). *Seed pots* are being manufactured from manures (CowPots). Manures can be used to generate *granular active carbon*, which can be used for water treatment (Lima and Marshall, 2005). Extracted proteins and amino acids from manures can be utilized as feed ingredients; also *black soldier fly* prepupae meal, which are raised on manures, can be a valuable feed ingredient (Bondari and Sheppard, 1981; Sheppard *et al.*, 1994; Meyers *et al.*, 2008). Carbohydrate material from manure can be utilized to make *biodegradable plastics* and other products (ABCNEWS.com, 2001). While there are many potential value–added products that can be generated from manure, none of these are currently in mainstream production.

Barriers to Manure Treatment Technology Adoption

Although there are a great number of commercially available systems for manure treatment and fractionation including solid–liquid separation, generation of biogas, nutrient extraction and value added products, these are still not common practices on most livestock farms. There are two main driving forces that prevent wide scale adoption of many manure treatment and fractionation technologies. The biggest deterrent to the adoption of many technologies is that they are not economically viable. While the technologies may be available, they do not produce enough value to make them attractive to most producers. This is due in part to cheap energy prices, relatively low costs of fertilizers, and a lack of demand for the different products derived from manure fractionation. Another large barrier to technology adoption is the on-farm management of some of these technologies. Most producers are focused on managing livestock production and do not have the time or interest to operate and maintain other equipment on site. For many of these technologies to be successfully adopted and operated, third-party collaborators will have to be involved that will be willing to install, operate and maintain the equipment on site without interfering with animal/farm management. Both of these hurdles will need to be overcome in the future in order for these innovative technologies to become common on the average livestock farm. One other issue of concern, in some cases, is the issue of scale. Some technologies are proven on a very small scale and would need to be scaled up to make them economical on larger livestock operations. On the other hand, for some technologies such as biogas production, there may need to be a larger source of feedstocks for the technology to be cost effective, which would eliminate the possibility of utilizing these technologies on smaller farms or would require a central production site that received manure from surrounding farms.

Future Research Needs

While many technologies for deriving more value from manure exist there needs to be more research done to improve the efficiencies of these technologies and make them more cost effective for the average producer. Some examples of areas needing further consideration are listed below, although this list is not exhaustive.

- Improve the N retention in composting practices.
- Improve solid–liquid separation and make it more cost effective.
- Improve thermal conversion technologies to make them cost effective.
- Improve anaerobic digestion technologies to make them more cost effective.
- Improve upon nutrient extraction technologies to improve economics.
- Take proven technologies and scale them up to make them economical on farm.
- Evaluate existing and new technologies for their potential to generate environmental credits (sale of C offset and nutrient credits).
- Develop new technologies to capture and concentrate nutrients.

In addition to research, success of many of these technologies would be dependent on support from government agencies or local communities in order to make them economically viable.

Pathogens and Veterinary Antibiotics in Livestock Manures

While historical concerns related to livestock production have traditionally focused on nutrient pollution of waters and air quality, increased awareness of zoonotic pathogens and veterinary pharmaceuticals in animal manures is now recognized as a public health concern (Venglovsky *et al.*, 2009). Domesticated livestock, as well as wildlife, harbour a variety of bacterial, viral and protozoal pathogens, some of which are endemic and cannot easily be eradicated (Sobsey *et al.*, 2006). As a result, the pathogens can be found in fresh animal manures at production facilities and off farm when inadequately treated manures are used as soil conditioners and fertilizers (US EPA, 2005). Some pathogens commonly found in cattle, swine and poultry manures are *Campylobacter* spp., *Escherichia coli* O157:H7, *Salmonella* spp., hepatitis E virus, *Cryptosporidium parvum* and *Giardia lamblia* (Kraus *et al.*, 2003). The titre of these zoonotic pathogens can exceed thousands per gram of faeces, with infection causing temporary illness or mortality, especially in high-risk individuals (Hutchison *et al.*, 2005a; Klein *et al.*, 2010; Létourneau *et al.*, 2010). Exposure of humans to pathogens can occur through occupational exposures, ingestion of contaminated food and water, or aerogenic routes (Matthews, 2006; Dungan, 2010).

Use of antimicrobials in livestock production may also intensify the resistance of pathogens to antibiotics, reducing the ability to treat infected individuals (Boxall *et al.*, 2003; Bahe *et al.*, 2006). In Europe, the USA and other countries, antibiotics are used therapeutically (high doses) in livestock production to treat specific diseases or sub-therapeutically (low doses) by incorporating into feed to improve growth efficiency (Sarmah *et al.*, 2006). It is also common practice to administer multiple classes of antibiotics to livestock simultaneously at the production facility (Song *et al.*, 2007). Because not all antibiotics are absorbed in the gut of animals, they are excreted via urine and faeces in unaltered form and as metabolites (Halling-Soresen *et al.*, 1998; Boxall *et al.*, 2004). It has been estimated that as much as 80% of orally ingested antibiotic can be excreted in urine and faeces (Elmund *et al.*, 1971; Levy, 1992; Halling-Sørensen *et al.*, 2002). Several classes of veterinary pharmaceuticals and antibiotics, including coccidiostats, ionophores, lincosamides, macrolides, sulfonamides and tetracyclines, have been detected in surface waters adjacent to livestock operations (Campagnolo *et al.*, 2002; Hao *et al.*, 2006; Song *et al.*, 2007). In addition, the practice of land-applying livestock manure for its fertilizer value provides for the introduction of veterinary antibiotics (VAs) over large areas in the environment, which have been detected in soils and waters worldwide (Hamscher *et al.*, 2002; Christian *et al.*, 2003). For more detailed discussion on antibiotics in animal agriculture and other emerging issues, see Chapter 18, this volume.

Fate of pathogens in manure

Prior to land application, manures are generally stockpiled, stored in lagoons or pits, or anaerobically digested or composted, all of which can influence the ultimate survival of pathogens (Sobsey *et al.*, 2006; Ziemer *et al.*, 2010). In untreated liquid manures, pathogens may persist for a long time depending upon the storage conditions and temperature, type of slurry and pathogen type. In general, low temperature, optimal moisture and solids content and no aeration have been shown to enhance the survival of pathogens in manures (Jones, 1976; Kudva *et al.*, 1998; Venglovsky *et al.*, 2006). For example, *E. coli* O157, *Salmonella, Listeria* and *Campylobacter* were shown to survive for up to 6 months in dairy manure slurries (2% and 7% DM); however, in manure heaps (both turned and unturned) where temperatures were >55°C, the pathogens survived less than a few days (Nicholson *et al.*, 2005). In inoculated sheep manure, Kudva and co-workers (1998) found that *E. coli* O157:H7 survived for 21 months under varying climactic conditions when not aerated, but only 4 months when aerated, with the difference being attributed to drying during aeration. In sheep and cattle faeces (and cattle slurries), *E. coli* O157:H7 survived the longest without aeration at temperatures <23°C. In swine slurry, viable *Salmonella* spp. were recovered up to 300 days when stored at 4°C, while at 37°C none was detected after 7 days (Arrus *et al.*, 2006).

In contrast to bacteria, protozoan parasites (e.g. *Cryptosporidium* and *Giardia*) can survive in livestock manures for an extended period, which is likely due to their ability to form cysts and oocysts. The *Cryptosporidium* oocyst can resist die-off over a wide temperature range while remaining infective (Fayer *et al.*, 2000). In unstirred swine slurries spiked with *Cryptosporidium* oocysts, a 1-log reduction in oocysts was calculated to occur at 270 and 345 days during cooler and warmer temperatures, respectively (Hutchison *et al.*, 2005b). In contrast, the survival of *Giardia* cysts is highly temperature dependent. In a mixed human and swine manure, 90% of *Giardia lamblia* cysts were non-viable at 130 and 4 days when the respective incubation temperatures were 5° and 25°C (Deng and Cliver, 1992a).

Viruses are obligate intracellular parasites that are unable to replicate outside their host and, as a result, their numbers do not increase once released into the environment. A variety of physical, chemical and biological factors are responsible for the stability of viruses in animal manure management and treatment systems. Virus survival in manures is likely influenced most by temperature, pH (very high or low), NH_3, microbial activity, aggregation (virus clumping), encapsulation or embedding in membranes or particles, and indirectly through solids association (Deng and Cliver, 1995a; Sobsey *et al.*, 2006). In various animal manures, D_{90} values (time, in days, required for a 90% reduction of virus titre) ranged from <7 days for herpesvirus to more than 180 days for rotavirus (Pesaro *et al.*, 1995). With a bovine parvovirus and porcine enterovirus, D_{90} values in animal manures were 200–300 days when the viruses were kept at 5°C (Srivastava and Lund, 1980; Lund and Nissen, 1983). In mixed swine and human waste, poliovirus type 1 was more stable at 14°C than at 21°C, with respective D_{90} values of 52 and 19 days (Deng and Cliver, 1992b). When dairy manure was mixed with human waste, D_{90} values for hepatitis A virus at 5°C were 35 days compared with 8 and 7 days at 25° and 37°C, respectively (Deng and Cliver, 1995b). As with bacteria and protozoan parasites, livestock manures represent a potential viral hazard when applied to agricultural land without being treated (Sobsey *et al.*, 2006).

Fate of pathogens in soil

It has been demonstrated that pathogenic bacteria survive longer when manures are immediately incorporated into soils than when left on the surface for some time (Hutchison *et al.*, 2004). Pathogens on the surface may be exposed to UV irradiation, temperature fluctuations and desiccation that can potentially decrease their ability to survive compared with soil-incorporated pathogens. In soils, however, indigenous soil microorganisms have been shown to increase the inactivation rate of pathogens (Dowe *et al.*, 1997; Jiang *et al.*, 2002). The pH and temperature of soil also influences the survival of bacteria, which is limited by low soil pH and higher temperatures (Gerba *et al.*, 1975). Soil texture may also enhance the survival of pathogens, as *E. coli* O157 was reported to survive up to 2 months longer in loam and clay soils than in sandy soil (Fenlon *et al.*, 2000).

In sandy and clay loam soils amended with various manures, *Campylobacter, E. coli* O157, *Listeria* and *Salmonella* were found to survive as long as a month or longer (Nicholson *et al.*, 2005). Similarly, Stanley *et al.* (1998) detected *Campylobacter* for up to 20 days after the application of contaminated dairy slurry, while Jones (1986) reported survival times for *Salmonella* up to 259 days in soils amended with animal faeces. In addition, it was found that *Salmonella* may persist in soils for a longer period in a viable non-culturable state, thus avoiding detection via use of traditional culture-based techniques (Turpin *et al.*, 1993). *Listeria* are ubiquitous in the rhizosphere, making them well adapted to survive for extended periods in soils (Van Renterghem *et al.*, 1991). In dairy manure-amended soil, *Listeria monocytogenes* survived for up to 43 and 14 days when incubated at 5° and 21°C, respectively (Jiang *et al.*, 2004). Dowe *et al.* (1997) showed that chicken manure promoted better growth of *L. monocytogenes* than did liquid hog manure, but only when the competitive bacterial flora was reduced by autoclaving.

In soil, *Giardia* cysts were inactivated after incubation for 1 week at −4° and 25°C; however, cysts were recoverable from soils for 2 months when maintained at 4°C (Ziemer *et al.*, 2010). *Cryptosporidium* oocysts are more environmentally resistant and remained infective >3 months in soil at −4° and 4°C, but at 25°C degradation

of the oocysts was accelerated and samples were infective for a shorter period (Olson *et al.*, 1999). While low and high temperatures definitively affect the survival and inactivation of oocysts, changes in soil moisture content had little or no effect on their inactivation (Jenkins *et al.*, 2002). In contrast, Kato *et al.* (2002) found that inactivation rates of oocysts were greater in dry soils than in moist and wet soils that were subjected to freeze–thaw cycles. Jenkins *et al.* (2002) also reported that soil texture may influence the inactivation of oocysts, but it could not be ruled out if this effect was related to soil pH. However, in a field study later conducted by Kato *et al.* (2004), oocyst inactivation could not be correlated with soil pH, moisture and organic matter content.

Removal of viruses in soils occurs largely by adsorption, with viruses surviving about as long as pathogenic bacteria (Gerba *et al.*, 1975; Gilbert *et al.*, 1976; Sobsey *et al.*, 1980). Hurst *et al.* (1980) found that the survival of enterovirus, rotavirus and bacteriophage in amended soils was influenced by temperature, soil moisture, presence of aerobic microorganisms, degree of adsorption, level of extractable P, exchangeable aluminium and soil pH. Overall, however, adsorption and temperature had the greatest effect on virus survival in soil, with virus survival decreasing with increasing temperature. At 37°C, no enterovirus infectivity was recovered from soil after 12 days, but at 4°C the virus persisted for at least 180 days (Yeager and O'Brien, 1979). Due to the adsorption of the virus by soils and influence of temperature on their survival, the land application of sewage effluent during warm and dry months has been documented as a viable disposal option to minimize the off-site transport and survival of viruses (Bitton *et al.*, 1984; Straub *et al.*, 1993).

Transport of pathogens in soil

Application of livestock manures on soils, particularly surface application of manure, can result in the transport of manure pathogens to surface or ground waters (Abu-Ashour *et al.*, 1994; Jamieson *et al.*, 2002; Tyrrel and Quinton, 2003). The overland transport of microorganisms is also called horizontal movement, while the leaching of microorganisms through soil and other porous subsurface strata is referred to as vertical movement. Unless a soil is saturated or contains an impermeable barrier, vertical movement of microorganisms will occur (Mawdsley *et al.*, 1995).

Despite the existence of bacterial, viral and protozoal pathogens in manures, few studies to date have examined their vertical and horizontal movement in soils under field conditions (Thurston-Enriquez *et al.*, 2005; Close *et al.*, 2010). As a result, knowledge of pathogen transport in soils has largely been inferred from studies of faecal indicator organism movement at grazed pastures, feedlots and manure-amended soils (Doran and Linn, 1979; Young *et al.*, 1980; Edwards *et al.*, 2000; Soupir *et al.*, 2006) or soil column or block studies amended with pathogen-containing livestock manure (Gagliardi and Karns, 2000; Davies *et al.*, 2004; Kuczynska *et al.*, 2005; Semenov *et al.*, 2009). Some physical and chemical properties that influence the vertical movement of microorganisms are soil type, water content and water flow, microbe and soil particle surface properties, cell motility, pH, plant roots, temperature, and presence of micro- and meso-faunal organisms (Mawdsley *et al.*, 1995; Unc and Goss, 2004).

Rapid horizontal transport of microorganisms to surface waters can occur when either the rainfall intensity exceeds the soil's infiltration rate or when the soil becomes so saturated that no rainfall can percolate (Tyrrel and Quinton, 2003). Factors that influence the level of microbiological contamination in runoff from agricultural lands are organism die-off rates, quantity and type of manure applied, sloping terrain, rainfall intensity and water infiltration rate (Evans and Owens, 1972; Doran and Linn, 1979; Baxter-Potter and Gilliland, 1988; Abu-Ashour and Lee, 2000; Jenkins *et al.*, 2006; Ramos *et al.*, 2006). Methods to mitigate the offsite transport of microorganisms in runoff from manure-amended soils and livestock feedlots include use of vegetative filter strips (Coyne *et al.*, 1995; Fajardo *et al.*, 2001) or vegetative treatment systems with a settling basin for solids collection and a vegetated area (Koelsch *et al.*, 2006; Berry *et al.*, 2007). Alternatively, livestock manures

could be treated (e.g. composted, anaerobically digested) prior to land application, thus reducing subsequent risks associated with pathogens (Lund *et al.*, 1996; Tiquia *et al.*, 1998).

Aerosolization of pathogens

Pathogens can potentially become aerosolized during the land application of liquid and solid manures, representing a potential risk if inhaled in sufficient quantities or ingested after deposition on food crops and fomites (Brooks *et al.*, 2004; Dungan, 2010). When bioaerosols are released from a source, they can be transported short or long distances, eventually being deposited (Brown and Hovmøller, 2002; Jones and Harrison, 2004). Unlike zoonotic agents in manures, soils and waters, aerosolized microorganisms are highly susceptible to meteorological factors such as relative humidity, solar irradiance and temperature (Cox and Wathes, 1995). In general, both laboratory and field studies have shown that the viability of aerosolized microorganisms decreases with decreases in relative humidity and increases in ambient temperature and solar irradiance (Mohr, 2007). Despite the potential for bioaerosol formation during these activities, very few studies have investigated the risk of human exposure to pathogens during the land application of animal manures.

During the land application of swine and cattle slurries by tanker and high-pressure spray guns, airborne bacterial counts steadily decreased with distance from the application site and pathogenic bacteria such as *Salmonella* spp. and *Klebsiella pneumoniae* were not detected (Boutin *et al.*, 1988). During the spray irrigation of swine slurry, a marker strain of *E. coli* was detected 125 m downwind in aerosols, but not at 250 and 500 m downwind (Hutchison *et al.*, 2008). Using polymerase chain reaction (PCR) to amplify 16S ribosomal RNA genes in air samples collected immediately adjacent to the spreading of swine and dairy cattle slurry, pathogens having an aerogenic route of infection were not identified (Murayama *et al.*, 2010; Dungan, 2012). While results from these and other studies suggest a low risk for exposure to pathogens (Brooks *et al.*, 2005; Tanner *et al.*, 2005), significant knowledge gaps still exist with respect to the fate and transport of bioaerosols, making it difficult to predicate the health risks associated with aerosolized pathogens accurately (Pillai and Ricke, 2002).

Fate, transport and negative impacts of veterinary antibiotics in soil

Once in the soil, antibiotics can be transported to surface and ground waters in a dissolved phase or sorbed to soil particles and colloids (Kay *et al.*, 2004; Song *et al.*, 2010). Tetracyclines have been shown to strongly sorb to soils, while macrolides, such as tylosin, have a weaker tendency to sorb (Rabølle and Spliid, 2000; Allaire *et al.*, 2006). In contrast, sulfonamides are likely the most mobile of the antibiotics and have been detected in groundwater at relatively high concentrations (Hamscher *et al.*, 2005; Batt *et al.*, 2006). Despite such knowledge, there is still little known about the occurrence, fate and transport of VAs in the soil–water environment. Recent research has shown that the addition of pig manure to soil caused a temporary increase in tetracycline resistance genes soon after manure application (Sengeløv *et al.*, 2003). When manures are land applied, resistant bacteria are also transferred, creating the possibility of horizontal transfer of resistance genes to the indigenous soil bacteria. The addition of nutrients to soils has been shown to enhance horizontal transfer to bacteria by providing nutrients for activation of transfer as well as mobilizing genetic elements (Top *et al.*, 1990; Heuer and Smalla, 2007). Furthermore, antibiotics and their metabolites in the manures might give resistant bacteria a selective advantage after being land applied (Halling-Sørensen *et al.*, 2001).

In addition to concerns over the proliferation of antibiotic-resistant bacteria, other concerns over antibiotics in the environment are related to negative impacts on water quality and soil microbial communities. While detectable levels of antibiotics have been observed in natural waters throughout the USA, much needs to be learned about the chronic effects of low-level exposures of pharmaceuticals on human and environmental health (Kolpin *et al.*, 2002; Focazio *et al.*, 2008). Currently, there

are no provisions within the Safe Drinking Water act to monitor or regulate antibiotics. With respect to impacts on soil microorganisms, sulfonamides were found to reduce enzymatic activity and have a prolonged effect on microbial community structure and diversity (Schmitt *et al.*, 2005; Thiele-Bruhn and Beck, 2005; Hammesfahr *et al.*, 2008; Gutiérrez *et al.*, 2010; Toth *et al.*, 2011). In contrast, chlorotetracycline was shown to have no effect on soil respiration and bacterial community structure, which can be attributed to sorption of the antibiotic to the soil matrix (Zielezny *et al.*, 2006). However, effects of sulfonamides were only observed when soils were amended with a C source (e.g. manure, glucose), which was responsible for stimulating bacterial growth and activity. This is an important implication, as use of manure as a fertilizer source may not only enhance the horizontal transfer of antibiotic resistance genes, but exacerbate the effect of antibiotics on the soil microbial community. As a result, there is great interest in understanding the effect of sustained applications of antibiotic-containing manures on the long-term health and function of agronomic soils. Of additional concern is the ability of some plant species to absorb antibiotics into their tissues, creating another route for exposure of humans to antibiotics (Kumar *et al.*, 2005).

Effect of manure treatment technologies on pathogens

To reduce risk factors associated with the land application of manures, various physical, chemical and biological treatment technologies could be used to reduce or eliminate the presence of pathogens (Heinonen-Tanski *et al.*, 2006). While there are advantages and disadvantages with these methods, some can provide additional benefits, such as the production of compost that can be used to enhance the properties of agricultural soils (Tester, 1990) or biogas for energy generation (Holm-Nielsen *et al.*, 2009). As mentioned previously in this chapter, there is a wide variety of technologies available to treat livestock manures; however, the only processes with a documented record of cost-effective pathogen reduction are composting and anaerobic digestion (Sobsey *et al.*, 2006; Martens and Böhm, 2009). Windrow composting was shown to eliminate *Salmonella* in pig manure when temperatures were maintained between 64° and 67°C for up to 3 weeks (Tiquia *et al.*, 1998). In a bench-scale study, *E. coli* O157:H7 and *Salmonella enteritidis* were not detected after 72 h of composting at 45°C (Lung *et al.*, 2001). In another laboratory study, Grewal *et al.* (2006) reported that *E. coli*, *Salmonella* and *Listeria* were not detectable after 3 days in dairy manure mixed with straw or sawdust and incubated at 55°C. When cattle slurry was fed into a mesophilic anaerobic digester for 24 days at an operating temperature of 28°C, only moderate reductions in *Salmonella typhimurium*, *Yersinia enterocolitica*, *L. monocytogenes* and *Campylobacter jejuni* were reported (Kearney *et al.*, 1993). Pathogenic reductions, however, are generally greater at higher temperatures used in thermophilic digesters (Lund *et al.*, 1996; Burtscher *et al.*, 1998). As an added benefit, anaerobic digestion and composting have also been found effective in significantly reducing the level of VAs in livestock manures (Arikan, 2008; Ramaswamy *et al.*, 2010; Wu *et al.*, 2011).

Summary

The sustainability of modern manure management is far from certain, with many demonstrating significant limitations from the standpoint of efficient use of manure resources and protection of environmental quality and human health. As demonstrated in this chapter, for manure management to be sustainable, a broad array of issues must be considered and addressed, all in the context of highly competitive modern livestock production systems that largely seek to minimize costs to the consumer. In the past decade, there have been major innovations in the areas of land application, manure treatment and processing and in the science of understanding the impact of manure management. As a result, major opportunities exist to improve the components of manure management. To be sustainable, these optimized components must work within the constraints of the broader livestock production system.

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8 Water-related Issues in Sustainability: Nitrogen and Phosphorus Management

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Introduction

Water quality in the USA and around the world is threatened by contamination with nutrients, primarily nitrogen (N) and phosphorus (P). Animal manure can be a valuable resource for farmers, providing nutrients, improving soil structure and increasing vegetative cover to reduce erosion potential. At the same time, application of manure nutrients in excess of crop requirements can result in environmental contamination.

Concentrated animal agriculture has been identified as a significant source of nutrient contamination of surface water, N contamination of groundwater and ammonia emission. Areas facing the dilemma of an economically important livestock industry concentrated in an environmentally sensitive area have few options. If agricultural practices continue as they have in the past, despite the significant changes in agricultural intensity and changing environmental conditions, continued damage to water resources and a loss of fishing and recreational activity are inevitable. If agricultural productivity is reduced, however, the maintenance of a stable farm economy, a viable rural economy and a reliable domestic food supply are seriously threatened. This chapter reviews impacts of nutrient pollution on ground and surface water, issues of nutrient imbalance related to animal agriculture in developing and developed countries, and impacts of more stringent regulations on water quality.

Water Quality Concerns

When N in manure or commercial fertilizer is applied to land in excess of crop uptake, contamination of groundwater via nitrate leaching may occur. Nitrate in drinking water is converted to nitrite in the human digestive tract. Nitrite can replace oxygen in haemoglobin, resulting in cyanosis, or oxygen starvation, especially in infants. The current EPA drinking water standard for nitrate N is 10 ppm (US EPA, 2002); all public water systems must comply. While private water supplies are not required to comply with this regulatory standard, a US Geological Survey report indicates that 7% of private wells are contaminated with nitrate (Hamilton *et al*., 2004). Nitrate concentrations exceeded standards in 20% of shallow wells in agricultural areas, as compared with 3% of wells in urban areas. Nitrate concentrations were highest in shallow groundwater beneath agricultural land use in areas with well drained soils (Burow *et al*., 2010). Alarmingly, nitrate concentrations in groundwater increased by 2 mg l^{-1} in the

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Delmarva Peninsula (Delaware, Maryland and Virginia) between 1988 and 2001, and the median concentration of nitrate in groundwater in this region exceeded the federal standard.

Excess N or P in water causes algae populations to grow rapidly, or to 'bloom'. The subsequent decomposition of the algae consumes dissolved oxygen in the water. Lack of dissolved oxygen is a major factor affecting the growth and reproduction of fish, clams, crabs, oysters and other aquatic animal life. An algae bloom and subsequent decrease in dissolved oxygen is known as *eutrophication*, and may be caused by runoff or leaching of P or N from land when application is in excess of crop requirements. Nitrogen is considered the first limiting nutrient for algae growth in saltwater systems, and P the first limiting nutrient in freshwater systems.

A classic series of whole-lake experiments established that P was the first limiting nutrient for eutrophication of freshwater (Schindler, 1977). Before this, small-scale bottle experiments had suggested that carbon may be the major limiting nutrient for eutrophication in some lakes, and the role of N in freshwater systems was unclear. But Schindler (1977) observed that phytoplankton growth was not compromised in lakes when less carbon was supplied by fertilizers; invasion of atmospheric CO_2 was sufficient to support normal growth of phytoplankton. Also, the growth of phytoplankton was not reduced when N supplementation was restricted; fixation of N by blue-green algae compensated. Instead, phytoplankton growth was proportional to total P concentration. The author concluded that a complex series of biological, geological and physical processes were adequate to maintain the N and C concentration in natural waters but that neither external nor internal mechanisms were evident to correct P concentration required for phytoplankton growth. This series of studies established firmly that nutrient management plans to limit eutrophication in freshwater systems should be based primarily on P.

Nutrient Imbalance with Intensive Animal Agriculture

Increased specialization and concentration of livestock and crop production has led to the net export of nutrients from major crop-producing areas to areas with a high concentration of animal agriculture. Livestock utilize N and P inefficiently, excreting 60–80% of that consumed. Therefore, the majority of nutrients brought on to the farm in feed stay on the farm rather than being exported in meat, eggs or milk. Animal manure is typically land-applied to supply nutrients for crop growth, but application in excess of crop needs results in nutrient losses and contamination of groundwater, surface water or air. More detail on manure application and sustainable use of manure in animal agriculture is given in Chapter 7, this volume.

Concentrated animal agriculture has been identified as a significant source of N and P contamination of surface water (median contribution 6.8–48% of P export, and 5.2–23% of N export depending on watershed; Smith and Alexander, 2000). The relative importance of different nutrient sources varies greatly in different regions. Animal agriculture is a minor source of nutrient pollution in the populous Northeast and Great Lakes regions of the USA, for instance. In contrast, the Shenandoah Valley of Virginia is an example of an area of intensive animal agriculture associated with increased contamination of surface water. The Shenandoah Valley has the highest population of both dairy cattle and poultry in the state, and as many as 20% of the dairy farms also have at least one poultry house. Estimated manure nutrient production in the Shenandoah Valley exceeds crop requirements on a yearly basis. Manure phosphate per acre of cropland increased by 90% between 1978 and 1992, and an analysis of soil tests in 2007–2010 indicated that nearly 90% of samples were ranked 'high' or 'very high' in P (Pease *et al.*, 1998). Although there is controversy as to the threshold level of soil test P that leads to P runoff, these soils clearly do not need additional P.

The link between animal numbers, manure application to a limited land area and P contamination of surface water was also demonstrated in the Lake Okeechobee watershed in Florida. From 1973 to 1988, P concentration in the water of Lake Okeechobee in Florida increased by 250% (Negahban *et al.*, 1993). During this same period, dairy cow numbers in the three counties surrounding the lake increased by more than 900 cows per year (Boggess *et al.*, 1997), and dairies were identified as the source

of 40% of the P load to the lake (Negahban *et al.*, 1993). The appearance of lake-wide algae blooms led to the imposition of stringent regulations designed to reduce agricultural runoff.

Increasing Regulatory Pressure

Increasing public concern about water quality and increased awareness of the potential impact of concentrated livestock production have led to the development and implementation of increasingly stringent environmental regulations. In the mid-1990s, with increasing animal numbers and changing animal production system, the need to minimize the impacts of Animal Feeding Operations (AFO) on water quality and public health led to increased federal attention to Concentrated Animal Feeding Operations (CAFO) regulations (USDA US EPA, 1999; USDA, 2003). The two primary regulatory approaches used in the USA are a permit approach focused on actual or likely polluters, and a more local, water-body-based approach known as Total Maximum Daily Load (TMDL) programme (USEPA, 1997, 1999, 2007, 2010; Walker, 2001).

The EPA directly manages the permit programme for CAFO in seven of the 50 states; the others have established their own regulatory programmes under EPA oversight. Permits are mandated for large CAFOs and for smaller operations demonstrating discharge. These permits include conditions such as Court decisions, which have made clear that the federal government can only require permits of farms actually discharging; size alone is not sufficient criteria. Many states, however, continue to determine permit eligibility primarily on size criteria.

Permit conditions vary by state. Effluent limits are mandated by 29 states while 37 states require nutrient management or land application plans. Some states require groundwater monitoring. Regular analysis of manure and soil, inspections and operator education are required to varying degrees. Nutrient and land application plans include best management practices (BMPs) to be implemented. The nutrient management plan is an enforceable part of the CAFO permit, so implementation of these BMPs is mandatory on permitted farms.

Metcalfe (2000) compared the indexes of animal manure management regulatory stringency of 19 states for 1994 and 1998, and found relatively low stringency (averaging 3) in 1994, increasing to 7 (in a 1–10 scale, 1 is the least stringent and 10 is the most stringent) for the year 1998. More stringent regulations will increase the costs of complying with new permit specifications. One negative of more stringent regulations is that it increases the appeal of violations. The more costly compliance, the greater the incentive to violators from avoidance of costs. Centner and Mullen (2002) suggested that more efforts and resources in enforcing the existing CAFO regulations may be more helpful achieving the goal of reducing pollutants than imposing additional regulations.

Nitrogen- versus phosphorus-based nutrient management plans

One key change in water quality regulations in the past 5 years is the shift from a primary focus on N to an increasing focus on P contamination of surface water. Most permit programmes have as their foundation nutrient management plans, farm-specific plans to manage the amount, form, placement and timing of the application of nutrients to crops, to provide for crop needs while minimizing the risk of nutrient losses. These plans traditionally focus on applying manure to meet the N needs of crops. When excreted, N and P are in imbalance in manure relative to crop needs. Land-application of manure to meet the N needs of the crop results in the over-application and accumulation of P in soils increasing the potential of P losses in runoff and subsequent eutrophication (Sharpley *et al.*, 2000).

Lander *et al.* (1998) estimated excess nutrients from animal feeding operations, defining excess nutrients as manure nutrients produced in excess of nutrient needs of local crop, hay and pasture land. Across the USA, 37 counties had excess manure N in 1992 but 137 counties had excess manure P. Using a different predictive approach, the USDA estimated that in 1997 there were 155 counties with surplus manure N from CAFOs compared with 337 counties with excess manure P (USDA, 2001). It is not clear if the increased number of counties with manure surplus from the 1992 study to the 1997 study reflects a real increase in nutrient surplus, differences in methodology or both.

Limiting manure application to the P needs of the crop is one way to avoid continued accumulation of P in soil and to minimize potential P runoff and contamination of surface water. Regulations limiting manure application to the P needs of the crop are in place for many states in the USA, and the federal CAFO regulations to address water pollution call for site-specific decisions on whether N- or P-based manure application limits are needed to protect water quality. Also, some federal cost-share funding is now being linked to the development and implementation of P-based nutrient management plans. Phosphorus-based nutrient management regulations dramatically increase the amount of land required to utilize manure and likely will have a severe, detrimental effect on much of the agricultural economy in areas of intensive animal agriculture. The areas with excess manure N or P are in critical need to develop and implement off-farm less expensive alternatives to land-application of manure.

Best management practices

The identification and implementation of solutions to the generation of excess manure in confined animal feeding operations are necessary to enable such agricultural operations to thrive in environmentally sensitive areas. BMPs are often cost-shared (subsidized by the government or private organizations) and are included in the nutrient management plan that is the core of most CAFO permits. The impacts of BMPs have been reviewed frequently (e.g. Terry, 1993; D'Arcy and Frost, 2001; Prokopy *et al.*, 2008); a few of these are briefly described below.

There is increasing recognition by industry groups that cattle should be excluded from direct access to streams and ponds because it is the right thing to do, both environmentally and economically. There is significant evidence that, if allowed access to streams, animals are a direct source of pollutants, and they reduce riparian vegetation, which makes them vulnerable to erosion and physical alteration of stream banks (Kauffman *et al.*, 1983; Williamson *et al.*, 1996; Wohl and Carline, 1996; Line *et al.*, 2000; Parkyn *et al.*, 2003; Ranganath *et al.*, 2009).

Stream fencing can refer to both the total exclusion of livestock from streams and the use of fences to restrict livestock to crossings and/or small watering areas. Either is highly effective in reducing pollution of streams. Two streams with extensive riparian grazing had a nearly complete lack of woody vegetation while the bank of a third stream with no grazing was vegetated with a mixture of grasses, shrubs and trees (Wohl and Carline, 1996). Of the ungrazed stream banks, 6% were eroded, but 81% of the stream banks were eroded in grazed areas. Subsequently, a combination of stream bank fencing, bank stabilization and installation of rock-lined animal crossings were implemented. These changes resulted in large reductions in total suspended solids and in fine substrates in the two streams that had been grazed. Exclusion of livestock from stream banks promoted rapid re-vegetation and stabilization of eroding areas. Reductions in sediment load were accompanied by increased populations of the insects (mayflies, stoneflies, beetles, bugs and flies), worms, snails, prawns and marron that serve as food sources for fish, as well as improvement in fish communities.

Reduced stream-bank erosion (Kauffman *et al.*, 1983) increased ground cover (Ranganath *et al.*, 2009), and reduced stream concentration of total N (Macklin, 2011) and total P (Macklin, 2011) are all reported with exclusion of livestock from streams. Soil and P loss were lower in pastures where cattle were prevented from stream-access by fencing than in grazed areas (6–61 versus 94–266 t km^{-1} $year^{-1}$ and 3–34 versus 37–122 kg km^{-1} $year^{-1}$, respectively; Zaimes *et al.*, 2008). However, fencing of every stream to exclude cattle may not be an economical strategy (Fitch and Adams, 1998). Fencing of riparian areas (the zone adjacent to a stream or any other water body) is expensive, with both installation and annual maintenance costs. In addition, the width and route of some streams fluctuate widely throughout the year as rainfall varies.

Questions about the economical and managerial feasibility of fencing cattle out of streams has led researchers to explore alternatives to stream bank fencing. Some research indicates that merely providing off-stream water sources will reduce time spent in streams even without fencing cattle out of streams (Miner *et al.*, 1992; Godwin and Miner, 1996; Sheffield *et al.*, 1997). When an off-stream watering area was made available, the time spent near the stream by four beef cows reduced from 60 to 15 min day^{-1} during

summer months (Godwin and Miner, 1996). Similarly, time spent by free-range cattle in or near the stream decreased by 86% when alternative water sources were provided between the feeding area and the stream and by 97% when the feeding area was located between the water tank and the stream (Miner *et al.*, 1992). Similar dramatic reductions in time spent in stream (Sheffield *et al.*, 1997), stream bank erosion (Sheffield *et al.*, 1997) and loading of solids, N and P (Sheffield *et al.*, 1997; Line *et al.*, 2000), have been observed by others with provision of alternative water sources.

Rotational grazing can also result in improved water quality, even without stream exclusion. Stream bank stability and in-stream habitat were similar between pastures with buffer strips and rotationally grazed pastures, and both were significantly improved over continuous grazing (Weigel *et al.*, 2000). Rotational grazing encouraged better sod development and helped stabilize erosive stream banks better than continuous pasture. Continuous pastures resulted in poor habitat quality and poor fish communities.

Total maximum daily loads

A significant evolution in US environmental regulations is the implementation of total maximum daily loads (TMDL) programmes. A TMDL is a goal, a target load of a specific pollutant that a specific stream segment can absorb on a daily basis without impairing a beneficial use (i.e. aquatic life, human recreational activities, human consumption, municipal uses; USDA, 2001). A TMDL is also a plan to restore impaired stream segments to a form that will support these beneficial uses. Finally, a TMDL is intended to be a process – identifying a goal for loading of a pollutant, developing a plan to meet that goal, implementing the plan, assessing the effectiveness of that plan, revising and re-implementing the plan (US EPA, 1991).

The concept of TMDLs was first laid out in the 1972 Clean Water Act. EPA was to direct the states to establish TMDLs for all streams and rivers, publish lists of impaired stream segments and develop plans for returning impaired waters to unimpaired status. Very little was done with this through the 1980s as the EPA focused on reducing pollution from point sources (e.g. industrial and municipal waste). The EPA became active regarding the development of TMDLs for non-point sources of pollution in response to lawsuits filed by environmental organizations in several states. In these lawsuits, environmental organizations alleged that EPA had failed to fulfil the requirements of the Clean Water Act as they had neither pushed states to develop TMDLs regarding non-point sources of pollution nor taken on responsibility for developing these TMDLs themselves.

Because of these lawsuits, guidelines for developing TMDLs were published in 1991, almost 20 years after the TMDL programme was mandated. Since then different approaches have been used to develop and implement TMDLs (Jesiek, 2007) including mass balance models, statistical methods and computer-based simulation models; the majority of states have chosen this last approach. The selection of models depends on the impairment, type of pollutant and complexity of the system.

The TMDL list identifies stream segments that are impaired with regard to pathogens, sediments, nutrients or benthic communities (aquatic wildlife). Development of a TMDL for a specific pollutant (i.e. coliform bacteria, nitrates, benthics) involves quantifying the capacity of a specific stream segment to incorporate that pollutant and adapt without negative impact. Predictive modelling is then done to quantify the allowable level of pollution for that stream segment. Sources of the pollutant are identified, and allowable pollution is allocated among sources. Sources include both point sources and non-point sources, and the allocation of the TMDL is to be specific (i.e. to a specific facility or farm).

In developing TMDLs, factors considered are the background sources of the pollutant (i.e. sources of pollution not related to human activity) and variation in loading of pollutants throughout the year. In developing plans to bring impaired waters into compliance with water quality goals, options include trying to minimize peaks in a highly variable pollutant load, or attempting to reduce all loadings by a specific percentage.

Once sources of pollution are identified, load reductions must be allocated to sources. Allocation of the load (i.e. who can continue to load pollutants to the stream and at what levels)

is obviously contentious. There is no single best method available to estimate load reduction allocation for multiple pollutants but most aim for uniformity in reducing load for all types of sources, and selection of sources with minimal regulation costs. Uncertainties in the model are acknowledged by inclusion of a margin of safety (MOS). Practices must be adopted by sources that will reduce pollutant loads to meet their assigned reductions. These practices include BMPs such as stream bank fencing, as well as tightened discharge limits for factories and wastewater treatment plants and improvements in septic systems of local homes.

Efficient implementation of TMDL plans demands stakeholders' active involvement. Early TMDLs suffered from lack of buy-in; increased efforts to elicit stakeholder input have resulted in more accurate assessment of sources of pollutants and more equitable allocation of load reductions. Effluent standards, subsidized BMPs, public education and effluent trading are among the tools used to restore water quality. The uncertainty in estimating MOS for N and P makes it almost impossible to set numeric criteria accurately for nutrient-based TMDLs. The lack of numeric set point for the nutrients may make the problem more aggravated but progress has been made in reducing nutrient pollution with TMDLs. The proportion of TMDLs focused on N and P decreased from 23.8% in 1996 to 7.7% in 2008 and 6.4% in 2012 (US EPA, 2012).

Do more stringent regulations result in improved water quality?

Despite accelerated regulatory activity since the mid-1990s, no consistent improvement has been observed in ground and surface water in the USA. Flow of total N decreased from a baseline period of 1980 to 1996 through subsequent moving 5-year average periods through 2006. Using an updated statistical model, nitrate N transport to the Gulf of Mexico increased by 9% between 1980 and 2008 (Sprague *et al.*, 2011). The rate of change was greater between 2000 and 2008 than in the 20 years previous. Flows of total P increased compared with baseline. Similarly, groundwater nitrate-N concentration increased significantly between samples obtained in 1988–1995 (495 wells across the USA) and samples from the same wells obtained during 2000–2004 (Rupert, 2008).

Denmark serves as a lesson for long-term effects of stringent policy measures intended to reduce N losses from farms. The agriculture industry in Denmark is particularly intense, and that country was early to realize the problems associated with N losses to water resources. With its relatively small population and agriculture industry, strict enforcement of regulations is possible. Beginning in 1985 and continuing for 20 years, Denmark enacted seven national action plans to reduce N pollution of water bodies. The instruments of these regulations were mandatory improvement in wastewater treatment plants, mandatory implementation of fertilizer and crop rotation plans to limit plant-available N application, and development of statutes codifying assumed plant available N proportion in manures.

Following implementation of these policies in Denmark, land application of manure N decreased from 244,000 to 237,000 t over 13 years, resulting in a 34% reduction in N surplus as measured by field balance (Kronvang *et al.*, 2008). In those 13 years, model-simulated N leaching (Danish DAISY model) from 86 small agricultural catchments representing eight georegions was reduced by 28% to 45%, annual total N concentration in streams draining in those same 86 catchments was lower in nitrate-N than the target concentration (50 mg l^{-1}) and annual N load was reduced by 32%. Thus, strict national regulations enabled Danish agriculture to reduce N leaching and N loads to surface waters while maintaining crop production and increasing livestock production.

The Water Framework Directive (WFD) was developed by the EU to collect information throughout the Europe in order to provide policy makers with the data to develop policies addressing water related issues (Collins and McGonigle, 2008). Three approaches were used: long-term water quality assessment by surveillance monitoring, identification of water-bodies failing to meet water quality standards, and understanding of the primary cause of failing to meet the standards. These data were used to develop efficient modelling and decision support tools.

A primary challenge with this and other monitoring and modelling efforts is the time lag between the implementation of a management

strategy and its impact on water quality. Reduction in surface loadings of nitrate does not instantly reduce its concentration at discharge points, and P management practices may take several years to improve water quality. Also, changes in sources of streamflow over time make the interpretation of water quality data difficult. The lack of improvement in ground and surface water quality in the USA is frustrating, but what is not knowable is what the changes would have been without the adoption of conservation practices promoted in the last two decades.

Also, the original EU models and many others fail to consider the risks from climate change, which has a complex relationship with diffuse pollution. For example, flooding may increase transport of sediment-associated pollutants but, also, drought may make aquatic ecosystems more vulnerable to various pollutants due to reduced dilution. Improvements in these types of predictive models should include factors such as future climate change and projected land use, so that policy makers can have different modelling options to fit specific situations.

Strategies to manage diffuse sources of pollution including agriculture must address the societal interest in minimizing the cost of management implementation while maximizing efficiency and effectiveness. Making farmers aware of peripheral benefits to reduced nutrient losses (e.g. reduced mastitis associated with stream exclusion) helps increase their willingness to pay a greater portion of the bill. Also, 'pollution swapping' must be avoided. Mitigation approaches for one pollutant may have negative effects on another. For instance, slurry application during spring rather than winter reduces nitrate in runoff but higher temperatures may increase ammonia emission. Similarly, composting of manure is positively perceived as it creates a stable, transportable, marketable product. The 'stability' of N in compost, however, is largely because most ammonia N is lost during the composting process.

Nutrient Imbalance, Water Pollution and Soil Nutrient Reserves in Developing Countries

While the nutrient balance challenge in developed countries is often one of excess, nutrient input inadequate to replenish soil is as often the problem in developing countries. In all countries nutrient losses from agricultural land is a non-point source of environmental degradation but in developing countries nutrient losses from croplands are more damaging still as they cause depletion of soil nutrients, impairing crop productivity. In both scenarios, deterioration of surface water is the outcome. Negative nutrient balances are observed in sub-Saharan Africa, making depletion of soil nutrients a continual threat (Saleem, 1998). Livestock manure is a key source of nutrients to replenish soil nutrients. Total N and P in animal excreta worldwide in 1996 were 93.7 and 21.1 million t, respectively, with cattle the dominant source for both N and P (Fig. 8.1).

Vitousek *et al.* (2009) compared nutrient imbalances in a low nutrient input maize-based system in Kenya, high nutrient input wheat- and maize double-cropping system in north-east China, and maize–soybean rotation system in upper Midwest USA (Table 8.1). In the Kenya system, annual N and P input to most fields was 7 and 8 kg ha^{-1}, which is insufficient to replenish the nutrients extracted by the crop (59 and 7 kg ha^{-1} annually). The opposite scenario was found in China where annual N and P input (588 and 92 kg ha^{-1}) was well in excess of the crop's requirement, leading to degradation of surface water degrading water quality. Similarly, during a period of 25 years (1970–1995), addition of N and P to agricultural fields in the Mississippi Basin far exceeded the amount extracted by crops leading to losses of nutrients to, and consequently eutrophication of, freshwaters and the coastal Gulf of Mexico. Nutrient N and P balances in this region are improving with annual N and P input averaging 155 and 14 kg ha^{-1}, and annual output of 145 and 23 kg ha^{-1} from 1997 to 2006, respectively.

Environmental policies must obviously be created based on local or regional systems. In the Kenya system studied, increased nutrient addition to the fields is clearly warranted. But the addition of N fertilizer can be reduced to half without affecting the yield or quality of the crop in China or in Mexico. In most cases, reduction of nutrient input alone will not be sufficient to restore water quality. Other management strategies are needed such as proper placement and timing of fertilization, dietary manipulation in

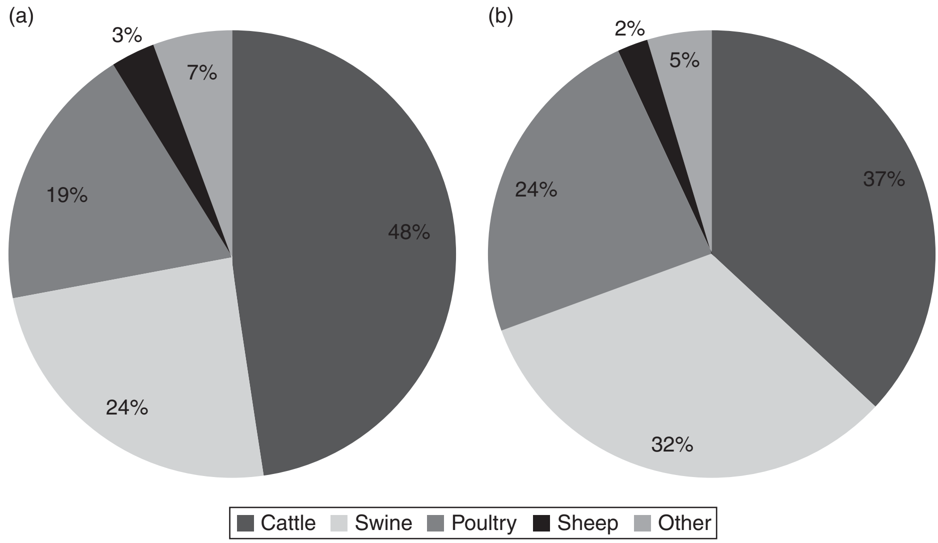

Fig. 8.1. Contribution of species of livestock to global production of (a) manure N and (b) manure P (adapted from Sheldrick *et al.*, 2003).

Table 8.1. Whole farm nutrient balance in different countries. Adapted from Vitousek *et al.* (2009) and Nkonya *et al.* (2005).

		Annual nutrient balance (kg ha^{-1})	
Country	Region	Nitrogen	Phosphorus
China	Northern	+227	+53
Kenya	Western	–52	+1
Uganda	Eastern	–48	–11
USA	Midwest	+10	–9

livestock farms, and protection or restoration of riparian vegetation. To maintain and improve the modern agriculture system, it is important for national agriculture agencies to evaluate the effects of changing system and to provide incentives to encourage the farmers to adopt nutrient-conserving practices.

Manure is one of the major contributors to nutrient inputs in all countries, and virtually the only contributor in some (Sheldrick *et al.*, 2003). Most countries, developed and developing, find manure a less expensive alternative to mineral fertilizers, but frequently commercial fertilizer is preferred because of its ready supply and easy application. From a global perspective, the use of manure as nutrient source has decreased with increasing use of synthetic fertilizers (Fig. 8.2).

Two countries with widely differing agriculture industries demonstrate changing fertilization practices and remaining opportunities (Sheldrick *et al.*, 2003). In Kenya, livestock production plays a major role in the agriculture industry, and since 1961 nutrient excretion by livestock has increased gradually. A relatively small portion of excreted N and P are captured in manure available for land application, suggesting great opportunity for improvement in productivity without reliance on additional commercial fertilizer. In the Netherlands, livestock manure has also been the key source of nutrient inputs but the majority of excreted nutrients are managed (contained, stored and applied in a managed fashion). Thus, they have been able to meet crop needs with reduced application of synthetic fertilizers.

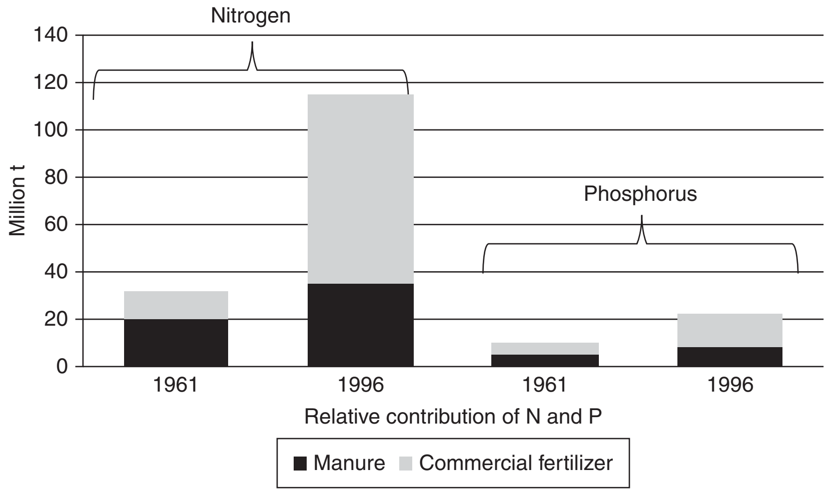

Fig. 8.2. Relative global use of manure and commercial sources of fertilizer in 1961 and 1996 (adapted from Sheldrick *et al*., 2003).

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9 Air Quality Issues in Sustainability: Greenhouse Gases and Volatile Organic Compounds

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Introduction

Animal agriculture production systems (beef, dairy, pork, etc.) play a critical role in the US food system, the economy, land use and nutrient cycling. Therefore, a thorough discussion of animal agriculture's sustainability requires examination of environmental, societal and economic components of sustainability. Within the environmental component, one area of interest is the carbonaceous and nitrogenous gaseous emissions that arise from animal agriculture production. For the purposes of this chapter, two broad classes of gaseous emissions will be discussed: greenhouse gas (GHG) emissions and volatile organic compounds (VOCs). Emissions of GHG are important to the discussion of sustainability because these gases contribute to climate change via their ability to absorb infrared radiation emanating from the surface of the Earth and emit radiation back towards the Earth's crust (Seinfeld and Pandis, 2006). The major GHG emissions from animal agriculture are carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O).

VOCs are a class of compounds that include thousands of gases and the US EPA defines VOCs as any carbon-containing compound that participates in atmospheric photochemical reactions, excluding carbon monoxide, CO_2, carbonic acids, metallic carbides or carbonates (EPA, 2011). These gases are important air pollutants because they contribute to the formation of ground-level ozone (Carter, 1994). This chapter will present a thorough discussion of the sources of GHG and VOC emissions from animal agriculture and potential emission mitigation techniques.

Sources of GHG and VOC Emissions from Animal Production

Greenhouse gases

GHGs include CO_2, N_2O, CH_4, water vapour and chlorofluorocarbons among others, and have the ability to absorb long-wave infrared radiation emanating from the Earth's crust and emit radiation back towards the surface of the Earth (Seinfeld and Pandis, 2006). As a result, GHGs play an important role in maintaining the global average temperature within a range that is conducive to life. It is estimated that without the

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'greenhouse effect' provided by GHGs the global mean temperature would be −19°C compared with the actual global mean temperature of 14°C (IPCC, 2007). However, since the beginning of the industrial revolution the atmospheric concentration of GHG has greatly increased (e.g. CO_2 concentrations have increased from about 280 parts per million (ppm) in pre-industrial times to 392 ppm in 2012; IPCC, 2007; NASA, 2012), and many scientists believe the increased atmospheric carbon concentrations are leading to global climate change (Oreskes, 2004). Ultimately, increases in atmospheric CO_2 concentration are being driven by the combustion of fossil fuels and land use change (e.g. clearing of forests) (Raupach *et al.*, 2007). Atmospheric concentrations of N_2O have increased from 270 parts per billion (ppb) in pre-industrial times to 319 ppb in 2005 due largely to increased fixation of atmospheric N_2 (which constitutes approximately 78% of the atmosphere) from human activities (e.g. synthetic fertilizer production) (Davidson, 2009). Atmospheric CH_4 concentrations have increased from about 700 ppb in pre-industrial times to around 1745 ppb today and anthropogenic sources of CH_4 include ruminant livestock, energy production, landfills and rice cultivation (Seinfeld and Pandis, 2006). The main GHG from animal agriculture production (CO_2, CH_4 and N_2O) have different global warming potentials (GWP) or abilities to trap heat based on their absorption spectrum and lifetimes within the atmosphere, so it is common to standardize all three to a 100-year CO_2 equivalent (CO_2-eq). The 100-year GWP of CO_2 is 1, CH_4 is 25 and N_2O is 298, which means CH_4 is 25 times more potent and N_2O 298 times more potent at trapping heat than CO_2 (IPCC, 2007).

There are many sources of GHG emissions from animal agriculture systems when considering the total production chain from the farm to dinner plate, including CO_2 and N_2O emissions from the soil, CO_2 emissions from the burning of fossil fuels (in farm equipment, transportation and electricity generation) and CH_4 emissions from manure management and enteric fermentation. Table 9.1 lists some of the sources of GHG from animal agriculture production. Carbon dioxide emissions also occur from the respiration processes of livestock, but this CO_2 is not considered a *net* emission because the animals are consuming plants that previously sequestered CO_2 from the atmosphere through photosynthesis (Pitesky *et al.*, 2009). For the purposes of this chapter, we will focus on the enteric GHG emissions from ruminant livestock and give an overview of the GHG from pig production systems.

Enteric CH_4 emissions refer to those that arise from the digestive tract of an animal as a result of microbial activities and represent a loss of dietary gross energy (GE) (up to 12% of dietary GE can be lost as CH_4 emissions in cattle; Johnson and Johnson, 1995). The majority of enteric CH_4 emissions result from microbial processes inside the rumen (the largest of four compartments comprising a ruminant's stomach) and not the hindgut (e.g. large intestine). However, though minor in comparison with those from the rumen, there are CH_4 emissions arising from the hindgut in ruminants (approximately 13% of total enteric CH_4; Ellis *et al.*, 2008). In comparison with ruminants, enteric CH_4 emissions from monogastric animals (e.g. pigs) are minimal and occur only from the hindgut.

Figure 9.1 shows a simplified model of the microbial processes that occur in the rumen that lead to CH_4 formation. When feed enters the rumen, it undergoes degradation and

Table 9.1 Three most important greenhouse gas emissions from animal agriculture production and their sources.

Greenhouse gas	Chemical formula	100 year global warming potential (IPCC, 2007)	Sources in animal agriculture production (cattle, small ruminants and pigs)
Carbon dioxide	CO_2	1	On-farm fossil fuel use, soil tillage
Methane	CH_4	25	Enteric fermentation (minor for pigs), anaerobically stored manure
Nitrous oxide	N_2O	298	Denitrification processes occurring in soil amended with manure

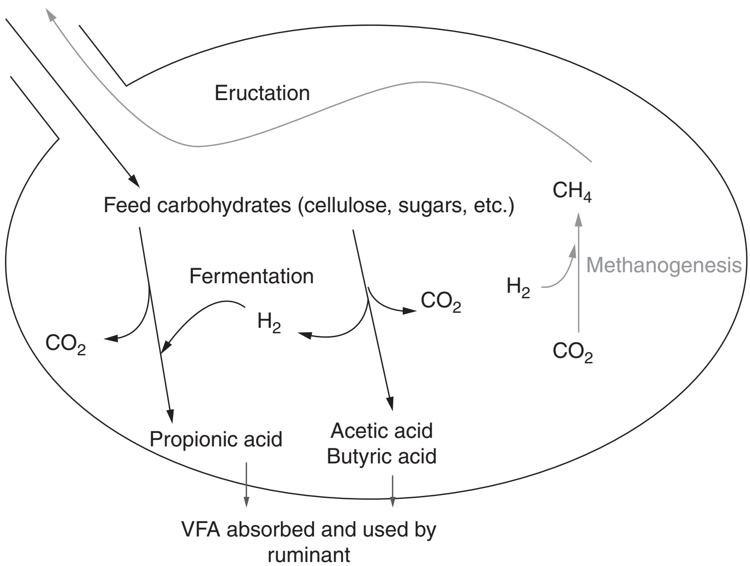

Fig 9.1. Simplified representation of the fermentation and methanogenic processes that occur in the rumen.

fermentation processes and the major resulting fermentation products are volatile fatty acids (VFAs). The main VFAs produced are propionic, acetic and butyric acid, which are absorbed by the animal through its rumen wall and can be oxidized for energy production, used for fat production and/or in the case of propionate, used for gluconeogenesis (production of glucose) (Bergman, 1990). Non-structural carbohydrates from plants (e.g. starch) tend to be fermented to propionic acid (though some dietary sugars are fermented to butyric acid), while structural carbohydrates (e.g. cellulose) tend to result in the production of acetic and butyric acid (Ellis *et al.*, 2008). When propionic acid is the end-product of fermentation there is a net use of hydrogen (H_2), while there is a net production of H_2 when acetic and butyric acid are the end-products of fermentation (Russell, 2002).

The microorganisms responsible for enteric CH_4 are called methanogens (classified within the Domain Archaea) and are strictly anaerobic (they require an oxygen-free environment to live) (Janssen and Kirs, 2008). Methanogens create CH_4 as an end-product of CO_2 reduction with H_2 in their electron transport chains to produce energy required for their life processes, though some methanogens may also reduce formate (Russell, 2002). The removal of H_2 by methanogens keeps the partial pressure of H_2 in the rumen low enough to prevent the inhibition of NADH-linked hydrogenases that would reduce the overall efficiency of rumen fermentation (Russell, 2002). Often methanogens are closely associated with the organisms that produce H_2, and ecto- and endosymbiotic relationships between methanogens and protozoa have been observed (Finlay *et al.*, 1994). Therefore, reducing the amount of H_2 available to methanogens or directly inhibiting methanogens or methanogenesis (methane formation) processes are the major ways to influence the total amount of CH_4 produced from enteric sources.

Nitrous oxide emissions from enteric fermentation are likely negligible and are often not included in estimates for national GHG inventories or life cycle assessments of emissions from each sector of animal agriculture (Monteny *et al.*, 2001; Casey and Holden, 2006). Kaspar and Tiedje (1981) found a small amount of N_2O originating from rumen fluid supplemented with nitrite and attributed it to dissimilatory nitrate reduction processes; however, the rumen fluid was incubated at temperatures below normal livestock body temperatures. Some studies measuring N_2O emissions from cattle in chambers have detected or quantified small amounts of N_2O, but those studies cannot separate emissions

from faeces and urine from enteric fermentation (Sun *et al.*, 2008; Hamilton *et al.*, 2010). Therefore, more definitive work is needed to determine whether N_2O emissions are originating from enteric fermentation processes and what mechanisms are responsible for these possible emissions.

Methane emissions from the large intestine of pigs is relatively minor, and are estimated to represent less than 1% of total digestible feed energy intake (Monteny *et al.*, 2001). The larger source of CH_4 emissions from pig production is manure management. Total CH_4 emissions from a Dutch pig production were estimated at 4.8 kg per pig per year with approximately 30% of those emissions coming from enteric fermentation and the balance from manure (Monteny *et al.*, 2006). Manure stored under anaerobic conditions can support methanogenesis, though the characteristics of the manure and the environmental conditions will greatly influence the total CH_4 emission. When higher concentrations of fermentable carbohydrates are fed to pigs, there is an increase in CH_4 emissions from anaerobically stored manure, as these diets increase the fermentable substrate available to microbial populations (Aarnink and Verstegen, 2007). Higher temperatures lead to greater metabolic activity in anaerobically stored manure and a higher rate of CH_4 emissions (Monteny *et al.*, 2006).

Nitrous oxide emissions from pig production, as in ruminants, primarily come from manure, not the animal directly. Denitrification and nitrification processes in manure and soil amended with manure are the primary source of N_2O emissions. Denitrification is a process performed by denitrifying bacteria where nitrate (NO_3^-) is converted to N_2 and N_2O is one of the intermediates produced during this process (Monteny *et al.*, 2001). Nitrification is the conversion of ammonium (NH_4^+) to NO_3^- and N_2O emissions are typically minimal during this process unless there is very little O_2 available (Monteny *et al.*, 2001). In addition to low O_2 conditions, higher N_2O emissions will result when easily degradable C-containing compounds are available (Velthof *et al.*, 2003).

Volatile organic compounds

As previously mentioned, VOCs contribute to the formation of 'ground level' or 'tropospheric' ozone (O_3; the troposphere is the lowest region of the atmosphere and where the majority of human activity occurs). Tropospheric O_3 is a major health concern and component of smog (Haagen-Smit, 1952). Ozone is one of the six 'criteria' pollutants currently regulated by the US EPA under the Clean Air Act. In the USA, there are no federal regulations for ambient VOC concentrations (the VOC content in consumer products such as paints is regulated by the EPA in an effort to improve indoor air quality), but air districts within the state of California have VOC emission regulations (EPA, 2011). Emissions of VOC can be broken down in to two groups: biogenic and anthropogenic. Biogenic emissions of VOC occur from 'natural' sources such as plants and the ocean (Seinfeld and Pandis, 2006). Anthropogenic VOC emissions are those that arise from human activities, and VOC emissions from animal agriculture fall into this category. How VOCs, also known as reactive organic gases (ROG), contribute to O_3 production is represented by this simplified general equation: VOCs+NO_x+sunlight-->O_3. In this equation, NO_x is an abbreviation for oxides of nitrogen, which includes nitrogen dioxide (NO_2) and nitric oxide (NO), and most NO_x emissions are a result of the burning of fossil fuels. Sunlight is a key driver of O_3 formation, and it is no coincidence that areas with abundant sunshine (and VOCs and NO_x) such as central and southern California and the Atlanta metro area often exceed EPA standards for O_3 (particularly in the summer months) (EPA, 2012). However, not all VOCs contribute equally to O_3 formation, and further examination of the role VOCs play in O_3 formation reveals why. Ozone is formed through what is known as the 'primary photolytic cycle'. Three reactions drive this cycle, and are as follows:

$$\mathrm{NO_2} + hv \xrightarrow{1} \mathrm{NO} + \mathrm{O}$$
$$\mathrm{O} + \mathrm{O_2} + \mathrm{M} \xrightarrow{2} \mathrm{O_3} + \mathrm{M}$$
$$\mathrm{O_3} + \mathrm{NO} \xrightarrow{3} \mathrm{NO_2} + \mathrm{O_2}$$

where NO_2 is nitrogen dioxide, *hv* is the energy from a photon, NO is nitric oxide, O is an oxygen atom, O_2 is molecular oxygen, M is a molecule (non-specific, but required to stabilize the formation of O_3) and O_3 is ozone (Knelson and Lee, 1977). Reaction 2 occurs very fast and Reaction 3 acts to replenish the supply of NO_2; therefore,

Reaction 1, or more specifically the concentration of NO_2, greatly determines the amount of O_3 produced (Seinfeld and Pandis, 2006). VOCs are not a part of these three reactions, but they play a critical role because they can oxidize NO to NO_2, thereby increasing the atmospheric concentration of NO_2 relative to NO. The ozone forming potential (OFP) of each VOC can vary widely, based in part on how the compound reacts in the atmosphere and the rate at which the compound undergoes the reaction(s) (Carter, 1994). Therefore, the OFP of a given VOC is important to consider when determining the environmental impact of its emission. Table 9.2 lists some of the common VOCs emitted from animal agriculture operations, their sources and the maximum incremental reactivity (MIR), which is a quantification of a compound's ability to form O_3 developed by Carter (1994). A larger MIR equates to a greater ability to drive O_3 production.

Recent research has revealed that fermented feeds (e.g. silage) seem to be a larger on-farm source of VOCs than manure (fresh or stored) as was previously assumed (Alanis *et al.*, 2008; Howard *et al.*, 2010 a). Feedstuffs, such as maize and lucerne, are often stored by farmers in a way to encourage their fermentation as a way to preserve those feeds for future use as a feed to livestock. Storage for these feeds includes stave silos, bunker silos and 'Ag Bags' (the feed is placed in long, air tight plastic tubes that are horizontal to the ground), and all systems are most effective at producing high-quality feeds when they are quickly filled and designed to minimize exposure to the air. Emissions of VOC from fermented feed are greater when the feed is exposed to air, greater wind speeds and higher temperatures (Montes *et al.*, 2010).

Experiments both on-farm and in environmental chambers have determined emissions of VOC can originate from the manure on swine and cattle production systems (Shaw *et al.*, 2007; Rumsey *et al.*, 2012). Blunden *et al.* (2005) collected air samples from sites downwind of swine farms in North Carolina and found that acetone, acetaldehyde, methanol and ethanol were the most prevalent VOCs found near the hog barns. Rumsey *et al.* (2012) measured VOC emissions from one North Carolina swine farm from both the barn and the manure lagoon over all four seasons. Lagoon VOC emissions

Table 9.2. Some of the common volatile organic compound species emitted from animal agriculture production and their mean incremental reactivity (MIR) from Carter (2009).

Compound name	Chemical formula	MIR	Livestock production sources	References
Acetic acid	CH_3CHOOH	0.66	Fermented feeds	Alanis *et al.*, 2008
			Fresh dairy cattle manure	Shaw *et al.*, 2007
			Enteric fermentation	Shaw *et al.*, 2007
Acetaldehyde	CH_3CHO	6.34	Swine manure lagoon	Rumsey *et al.*, 2012
			Swine barn	Blunden *et al.*, 2005; Rumsey *et al.*, 2012
			Fermented feeds	Howard *et al.*, 2010a
Ethanol	CH_3CH_2OH	1.45	Dairy manure lagoon	Filipy *et al.*, 2006
			Fermented feeds	Howard *et al.*, 2010a
			Dairy cattle housing area	Filipy *et al.*, 2006; Chung *et al.*, 2010
			Swine manure lagoon	Rumsey *et al.*, 2012
			Swine barn	Blunden *et al.*, 2005; Rumsey *et al.*, 2012
Methanol	CH_2OH	0.65	Enteric fermentation	Shaw *et al.*, 2007
			Fresh dairy cattle manure	Shaw *et al.*, 2007
			Swine manure lagoon	Blunden *et al.*, 2005; Rumsey *et al.*, 2012
			Swine barn	Rumsey *et al.*, 2012
Acetone	CH_3COCH_3	0.35	Dairy manure lagoon	Filipy *et al.*, 2006; Chung *et al.*, 2010
			Dairy cattle housing area	Filipy *et al.*, 2006; Chung *et al.*, 2010
			Swine manure lagoon	Rumsey *et al.*, 2012
			Swine barn	Blunden *et al.*, 2005; Rumsey *et al.*, 2012

were found to be highest in the summer months and lowest in the winter and spring, while barn emissions were found to be the highest in the spring and autumn and lowest in the winter and spring (Rumsey *et al.*, 2012). The study was not exhaustive enough to determine the mechanisms behind the seasonal variation. Chung *et al.* (2010) measured VOC emissions from six dairies in the San Joaquin Valley of California and found VOC fluxes from alleyways in free stall barns were significantly lower after flushing (a method of cleaning and removing the manure from the barns) than before flushing. Wind velocity over the surface of manure is another important factor contributing to the emission rate of VOCs, with higher wind velocities corresponding to higher emission rates (Parker *et al.*, 2010). In summary, VOC emissions from animal manure seem to be influenced by environmental factors and manure management techniques, though further research is needed to better understand the drivers of fresh and stored VOC emission variation.

Mitigation Options

GHG emission mitigation opportunities

Emissions of GHG across all farms within each sector of animal agriculture (e.g. beef cattle production, dairy production, swine production) vary widely due to differences in genetics, feeding regimes and management across farms. As a result, using generalized emission factors per animal do not accurately reflect the emissions from individual animal farming operations and mitigation strategies should seek to reduce CO_2-eq emissions per unit of output (kg of milk, kg or pork) rather than CO_2-eq emissions per animal. Thus, the overview of GHG mitigation strategies outlined below will be framed in the context of their ability to reduce CO_2-eq emissions per unit of output.

Mitigating CH_4 emissions from enteric fermentation directly requires reducing the output of methanogens residing in the gastrointestinal tract, which can be achieved by reducing the supply of CH_4 precursors, inhibiting the methanogenesis process and/or reducing or eliminating methanogen populations. The following enteric CH_4 emission mitigation strategies will be discussed: fats, forage quality/forage-to-concentrate ration, ionophores, alternative H_2 sinks and plant compounds.

Fats (specifically unsaturated fatty acids) can reduce enteric CH_4 emissions in two ways: acting as a sink for H_2 during biohydrogenation and causing toxicity to H_2-producing organisms. When unsaturated fats (those that contain double carbon-to-carbon bonds) enter the rumen, rumen bacteria perform a process called biohydrogenation, during which they saturate the double bonds by using H_2. Therefore, biohydrogenation processes can act a competitor for H_2 with methanogens and have the potential to reduce CH_4 production in the rumen (Ellis *et al.*, 2008). Additionally, unsaturated fats can be toxic to H_2-producing protozoa that often have endo- and ectosymbiotic relationships with methanogens (Hegarty, 1999). More unsaturated fatty acids have been found to have a greater toxic effect on protozoa and some Gram-positive bacteria, which is proposed to be the result of their ability to disrupt plasma membranes (Maia *et al.*, 2007). Research feeding ruminants fats has shown variable effects on CH_4 emissions that is believed to be due in part to the fatty acid profile of the fat source and the concentration at which the fats are fed (Beauchemin *et al.*, 2008, 2009). The concentration of fat in the diet is an important consideration because if too much fat is fed (over 5–6% of dry matter intake (DMI) for dairy cattle) rumen fermentation processes can be negatively impacted and DMI and animal performance can be deleteriously affected (Martin *et al.*, 2008). Additionally, the cost and contribution of sourcing some of the fats fed in experimental diets (e.g. coconut oil) should be considered in the total GHG emissions from livestock systems.

Forage quality and the forage-to-concentrate ratio can greatly influence CH_4 emissions from ruminants as diets that contain more structural carbohydrates tend to produce greater amounts of acetic acid as an end-product, and subsequently more H_2 and CH_4 than those diets that contain more non-structural carbohydrates (Ellis *et al.*, 2008). Consequently, the high concentrate diets fed to cattle in feedlots tend to result in fewer CH_4 emissions per unit of feed compared with higher fibre diets such as those consumed by dairy cattle and ruminants grazing

on rangelands and pastures. Aguerre *et al.* (2011) fed dairy cows increasing amounts of forage relative to concentrate and found CH_4 emissions increased from 538 to 648 g per cow per day for cows fed a 47:53 versus a 68:32 forage-to-concentrate diet. While feeding higher fermentable carbohydrate (e.g. maize grain) diets can produce lower CH_4 emissions, they are typically more expensive and can lead to rumen acidosis and a reduction of the animal's productive life, which makes those diets more typical for 'finishing' cattle and sheep (Owens *et al.*, 1998). Furthermore, feeding all ruminants high concentrate diets minimizes the unique attribute these animals have when compared with monogastrics: they convert the most abundant organic molecule on Earth, cellulose, into high-quality food and fibre products (Oltjen and Beckett, 1996).

Ionophores are antimicrobials fed to livestock that can transport ions across plasma membranes and include the compounds monensin and lasalocid. Both of these compounds were originally used as coccidiostats (to inhibit coccidiosis) in poultry and later cattle, but they have also been found to increase feed efficiency (Tedeschi *et al.*, 2003; Chapman *et al.*, 2010). The mode of action of ionophores for improved feed efficiency is they can transport cations (such as Na^+) across the plasma membranes of Gram-positive bacteria (which tend to be fibre-fermenting bacteria that produce acetic acid), thereby causing those bacteria to expend energy on pumping out the cations rather than on growth (Russell and Houlihan, 2003). Thus, there is a net shift from Gram-positive to Gram-negative bacteria within the rumen resulting in a lower acetic-to-propionic acid ratio and decreased CH_4 emissions, as less H_2 is available for methanogenesis (Tedeschi *et al.*, 2003). However, ionophore research with dairy and beef cattle has not always shown consistent reductions in CH_4 emissions, and possible explanations for the variable results include the ionophore dose, the time it is included in the diet (e.g. rumen microbes adapt to the ionophore over time) and the diet composition (Guan *et al.*, 2006; Hamilton *et al.*, 2010). Further exploration of the inconsistencies of ionophores needs to be completed before these compounds can definitively be proposed as a viable enteric CH_4 emission mitigation technique for ruminants. Additionally, past research has found that up to 50% of monensin can pass through the digestive tract intact in steers (Donoho *et al.*, 1978), and Varel and Hashimoto (1981) found that CH_4 production from anaerobically stored manure was inhibited from cattle fed monensin. This could be a beneficial, or in the case of manure CH_4 biodigesters where CH_4 production is desired, an unbeneficial side effect of feeding cattle ionophores.

Alternative H_2 sinks (e.g. sulfates and nitrates) are another enteric CH_4 emission mitigation strategy that has received more research attention in recent years. Sulfate-reducing bacteria utilize H_2 to reduce sulfate and other oxidized sulfur compounds, and in an environment where sulfate is not limiting can outcompete methanogens for H_2 (Ellis *et al.*, 2008). However, feeding diets too high in sulfate can depress DMI and cause negative health effects, so its use in reducing CH_4 emissions has been limited so far, though the increased prevalence of sulfur-containing distiller's grains from ethanol production may contribute to decreased CH_4 emissions from ruminants (Ellis *et al.*, 2008). Nitrates in the feed are first rapidly converted to nitrite, which can be detrimental to the animal if it accumulates at too high of a level (it can convert haemoglobin to methaemoglobin, preventing O_2 transportation throughout the body) (van Zijderveld *et al.*, 2010). However, research has shown that slowing 'stepping up' the amount of NO_3^- in the diet can successfully reduce CH_4 emissions in sheep and cattle without causing any negative health effects (van Zijderveld *et al.*, 2010, 2011). It remains to be seen if this mitigation strategy would be viable on-farm, as the economic cost and risk of health effects may be too high for some farmers to accept. Acetogens (organisms that reduce CO_2 with H_2 to acetic acid) can be found in ruminants and represent an ideal H_2 competitor because acetic acid can be used by the animal as an energy source; however, acetogens cannot compete as well as methanogens for H_2, which makes establishing dominant populations in the rumen difficult (McAllister and Newbold, 2008).

Plant-derived compounds such as tannins, essential oils and saponins are being increasingly researched as 'natural' alternatives to antimicrobials like ionophores. Essential oils (e.g. cinnamaldehyde and garlic oil) are so-named in reference to their odour, not that

they are required in the diet such as essential amino acids (Calsamiglia *et al.*, 2007). Much of the work with these compounds has been done *in vitro*, though a few studies have demonstrated reduced enteric CH_4 emissions *in vivo*. Mao *et al.* (2010) fed lambs tea saponins and found reduced CH_4 emissions and protozoa populations relative to total bacteria, but the relative number of methanogens was not affected nor was the lamb's growth. Mohammed *et al.* (2004) fed steers horseradish oil and found reduced CH_4 emissions, but Beauchemin and McGinn (2006) fed steers a commercial mixture of essential oils and found no effect on CH_4 emissions. Clearly, more *in vivo* research is needed to understand the effectiveness of various plant extracts, the doses required to impact CH_4 emissions and if there is an adaptation to the extracts over time. Furthermore, research is needed to evaluate their economic cost to farmers and the effects of such compounds on milk production and composition.

Mitigating CO_2-eq emissions per unit of pork can be achieved by reducing emissions from manure and by improving production efficiency (enteric CH_4 emissions from pigs will be ignored as they are relatively minor). Anaerobic digestion of manure can generate CH_4 gas that can be captured for energy production use on-farm and reduce the CO_2-eq emissions from swine production (Schils *et al.*, 2007). Cooling of manure slurry is another way to reduce CH_4 emissions (Schils *et al.*, 2007); however, this may not be cost effective for swine farmers. Nitrous oxide emission mitigation is best achieved through optimizing when manure is applied to soil, as it is important to time the application with plant's N needs for growth to minimize the nitrification/denitrification processes of soil microbes (Dosch and Gutser, 1996).

Chapters 2 and 3 (this volume) cover production efficiency more extensively, however it is worth mentioning as a proven method of reducing CO_2-eq emissions per unit of output (Capper, 2011; Gerber *et al.*, 2011). Production efficiency can be defined as achieving greater or the same output (e.g. kg of pork) with fewer inputs. Two of the ways to improve the production efficiency of pork production (and likely reduce emissions per unit of pork, though the authors are unaware of any published analysis) is to increase the litter size per sow and reduce the incidence of disease. Litter sizes in the USA have already made impressive historical gains (the average litter rate increased from around 8.8 in 2002 to just over 10 pigs per litter in 2011), and further improvements will likely reduce the number of sows required to produce each kg of pork, thereby reducing manure and CO_2-eq emissions from manure per unit of pork as well (NASS, 2011). One of the major diseases that impacts pigs is porcine reproductive and respiratory syndrome (PRRS). The PRRS virus can cause reproductive failure, pneumonia, reduced growth rates and mortalities in pigs (Neumann *et al.*, 2005). It is estimated the disease causes US$560 million in losses to the US swine industry each year, and, since the disease's emergence in the 1980s, effective control measures have yet to be found (Neumann *et al.*, 2005). It seems likely that finding effective treatments for diseases like PRRS would not only improve the welfare of pigs, but also reduce the inputs required and emissions created per unit of pork through improved production efficiency.

VOC emission mitigation opportunities

Mitigating the VOC emissions from fermented feeds can be achieved through using best management practices (BMPs) that are advocated for reducing feed losses and improving feed quality. Ultimately, these BMPs are focused around achieving optimum fermentation processes (for feed preservation), minimizing the exposure of feed to the air and reducing overall feed waste. Filling silos with harvested plant material quickly, harvesting the crop at the proper moisture content, packing the harvested feed adequately, using microbial inoculants and preservation acids, and completely sealing the feed to prevent air exposure can all promote optimum fermentation and preservation of silage (Muck, 1988). Most of the VOC emissions occur once the silo is opened for feed out, as the numerous VOCs in the aqueous phase of the silage (e.g. ethanol) will rapidly volatilize once exposed to the air. Recent research has shown that VOC emissions are far greater from 'loose' silage and most of these emissions occur within the first 12 h of air exposure (Hafner *et al.*, 2010). Therefore, the best way to reduce VOC emissions at feed out is to minimize the size of the 'face' of

the silo (the area that is exposed to air) by not opening more of the silo than is required and keeping the face smooth to reduce the surface area. Achieving this BMP goal will require proper planning of the silo design and the proper equipment to achieve a smooth face. Additionally, precision feeding (i.e. managing a livestock nutritional programme to precisely match an animal's nutrient requirements based on their stage-of-life) that requires tracking animal DMI and the dry matter content of the feed can minimize extra silage being fed to animals when it is not needed. This could potentially reduce VOC emissions by minimizing the exposed surface area of loose silage in feed alleys on farms. Other innovative ways to prevent or minimize VOC emissions from fermented feeds will undoubtedly be put forth in the coming years, and all will require experimental testing. Current and future solutions have the benefit of not only preventing emissions of a smog-precursor, but also minimizing costly losses of feed dry matter and nutrients for farmers.

Water-soluble VOC emissions (e.g. the alcohols, ethanol and methanol) could be mitigated from animal housing floors by flushing the manure with water as was demonstrated by Chung *et al.* (2010). However, consideration has to be given to the VOC emissions from the end point of this manure slurry (typical a lagoon) and the water-use required for flushing animal housing systems, which may be undesirable in arid locations like California. Biofiltration systems are another possible VOC mitigation technique for ruminant and swine production systems. A biofilter consists of a filter material (e.g. wood chips, soil) that contains microorganisms and can 'treat' the exhaust air from housing or manure storage system (Martens *et al.*, 2001; Pagans *et al.*, 2007). Research has shown biofilters can effectively reduce VOCs from swine housing facilities (Martens *et al.*, 2001); however, wide-scale adoption of biofilters would require demonstration of the environmental benefits and economic costs to farmers.

Measurement and Modelling of Emissions

Central to both the estimation of 'baseline' emissions inventories, or the testing of GHG or VOC emission mitigation strategies, is the acquisition of accurate and precise emissions data. Thus, capturing emissions from live animals, animal housing systems and feed management systems becomes essential. Having accurate and precise data is also required for validation of models that simulate or predict gaseous emissions from animal agriculture production. Measurement of GHG and VOC emissions from animals, their waste and feedstuffs can be measured in chamber systems or under field conditions. An overview and description of the enteric CH_4 emission measurement techniques for cattle can be found in Johnson and Johnson (1995). These methods include whole animal chambers, ventilated hood systems, the sulfur hexafluoride technique (a tracer gas method) and micrometeorological techniques (Johnson and Johnson, 1995). There are benefits and trade-offs with all of these techniques, with chamber methods generally collecting more precise data per animal than tracer or micrometeorological methods; however, chambers are often more restricting of the animal's natural behaviours (and normal emissions) than the other less restrictive methods. For further information on measurement techniques of GHG and ammonia, please see Chapter 15, this volume.

VOC emissions measurements from feed, animals and their waste have been completed in environmental chambers (Shaw *et al.*, 2007; Sun *et al.*, 2008), with transportable 'smog' chambers (Howard *et al.*, 2010a,b), and some on-farm VOC measurements have used isolation flux chambers, which limit the airflow over the measured surfaces and likely do not accurately represent emissions under 'normal' conditions (Montes *et al.*, 2010). Proper validation for all of these various GHG and VOC emission capturing systems and their gas analyser systems with known standards is paramount to ensure the data are accurate. Furthermore, collecting accurate emissions datasets is required to develop and improve existing models that can predict or simulate emissions from animal agriculture. The numerous existing emissions models and the strategies are beyond the scope of this chapter but have been reviewed by Archibeque *et al.* (2012). However, it is important to note that our current understanding of all of the bio-geochemical processes that lead to GHG and VOC emissions is limited, and

improving our understanding is necessary to advance the sustainability of animal agriculture production systems.

Conclusion

Air quality is a critical component of sustainable animal agriculture systems. GHG and VOC emissions from animal production systems can arise from the animals directly, their manure and from their feeds. Methane emissions from enteric fermentation and manure stored under anaerobic conditions result from microbial processes. Nitrous oxide emissions mostly come from nitrification and denitrification bacterial processes; therefore, direct mitigation of GHG emissions requires manipulation of microbial populations or their environment. Reporting GHG emissions per unit of output (e.g. per kg of milk or pork) is more appropriate than reporting per animal unit due the variability across farms. Improving production efficiency is a powerful and proven method to reduce GHG emissions per unit of output from animal production systems. VOCs are precursors to tropospheric O_3; however, not all VOCs contribute equally to O_3 formation, which needs to be considered when evaluating the sustainability of animal agriculture production systems. The major source of VOC emissions on farms is fermented feeds and to a lesser extent fresh manure. Mitigation of VOCs from fermented feeds should also reduce the loss of feed dry matter, and improve the economic profitability of the feeding enterprise on farms. In summary, reducing animal agriculture's impact on air quality will contribute to the industry's long-term environmental sustainability.

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10 Integration of Air and Water Quality Issues

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Introduction

As discussed in more detail in other chapters of this volume (Chapters 7, 8 and 9), there are many environmental issues of concern in animal agriculture. These include the effects of production on both air and water quality. There are a number of gaseous emissions from animal production with the major sources being the animal, manure excreted by the animal and feed (primarily silages) consumed by the animal. Compounds of concern include ammonia, hydrogen sulfide, volatile organic compounds and the greenhouse gases of methane, nitrous oxide and carbon dioxide. Emissions of ammonia and hydrogen sulfide can contribute to the formation of fine particulate matter in the atmosphere, which contributes to human respiratory problems (NRC, 2003). Ammonia emission contributes to regional degradation of terrestrial and aquatic ecosystems, and both compounds contribute to the local nuisance of odour (NRC, 2003; Renard *et al.*, 2004). Volatile organic compounds include many different alcohol, acid and aromatic compounds. In addition to their local effect of odour, they contribute to the formation of ground level ozone or smog (Howard *et al.*, 2010). Greenhouse gases are a concern due to their role in global climate change (IPCC, 2007).

Water quality effects of animal production primarily occur during feed production where nutrients and other compounds are leached through the soil to groundwater or carried through runoff to surface waters. Other contamination sources include runoff from feedlots and accidental leakage of manure into streams and other water bodies. For agricultural systems, nitrogen (N) and phosphorus (P) are the nutrient losses of most concern. Nitrogen, primarily in the form of nitrate, can leach to groundwater where levels greater than 10 ppm of N are a human health concern (EPA, 2009). Small amounts of nitrate can also drain off field surfaces into streams, rivers, lakes and large water bodies such as the Chesapeake Bay and Gulf of Mexico contributing to eutrophication and to death or elimination of many natural organisms (Conley *et al.*, 2009). Phosphorus can also leach to groundwater, but this is typically not a concern unless that groundwater moves back to the surface. Surface loss of P through runoff also contributes to the eutrophication of water bodies (Sharpley *et al.*, 2003).

Managing farms to reduce or control all of the different nutrient losses affecting air and water quality is very complex. There are many interactions among the farm components so reducing one source or type of emission may

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exacerbate others. For example, reducing ammonia emissions from manure leads to greater concentrations of N in manure during long-term storage and field application, which can cause greater nitrate leaching losses to groundwater and greater nitrous oxide formation and emission to the atmosphere (Rotz *et al.*, 2011c). Simultaneous reduction of all environmental impacts requires comprehensive evaluation and management of all aspects of the farm.

Another important aspect of farm management is profitability for the producer. As farm management or technological changes are made to improve environmental sustainability of the farm, this must be done with minimal cost, and preferably increased benefit, to the producer. Most animal-producing farms operate with a relatively tight profit margin, which is unable to absorb an increased cost of production. If production cost is increased by efforts to reduce losses to the environment, then this cost must be passed on through an increase in the price of the product or the producer must receive a government subsidy to maintain profit. The ideal situation is to reduce production costs sufficiently through improved nutrient use efficiency to cover any potential cost increases of management changes. Although such combinations can sometimes be found (Ghebremichael *et al.*, 2007), more commonly improvements in environmental sustainability come with increased production costs through new technology or increased labour and other resource requirements (Rotz *et al.*, 2006; del Prado *et al.*, 2010).

Improving the sustainability of animal production systems requires a comprehensive evaluation that considers all environmental issues along with profitability. This type of comprehensive evaluation is best done through process-level modelling and simulation of farming systems. Well-developed models can integrate the many processes involved in farm production, nutrient management and economics providing a more holistic evaluation of sustainability. Considering costs, benefits and tradeoffs throughout the system provides better informed decisions on the implications of a set of management changes than does considering only individual system components. This type of system-wide analysis provides a robust, quantitative basis for decision making and planning of future production strategies.

Whole-farm Modelling

Many models have been developed and applied to the evaluation of farming systems. Most of these have focused on the performance and economics of production systems. In the past decade, a number of farm models have also included an environmental component, estimating one or more forms of pollution created by the farming system. The goal in many evaluations is to reduce the environmental impact of the production system while maintaining or increasing farm profit.

Modelling techniques

Farm system models can be categorized into three major types: mathematical optimization, simulation and life cycle assessment (LCA). Although they may be used with similar goals, they are different in structure and function. Optimization programs work by simultaneously solving a series of linear or non-linear equations to obtain an optimum solution. This type of model has been widely used to determine economically optimal strategies for various farm production systems. A few studies have combined environmental and economic evaluation. For example, Berentsen and Giesen (1995) developed a deterministic static linear programming model of a dairy farm, which they used to analyse institutional and technical change in dairy farming. Effects of farm management on N losses were evaluated while optimizing farm profitability. Annetts and Audsley (2002) developed a multiple objective linear program focused on the environmental and economic planning of farm systems in the UK. The model, known as the Silsoe Whole Farm Model, focused on arable crop farming but included a livestock component.

Simulation-based farm models mimic major farm processes to represent the production system statically or dynamically through time. The Moorepark Dairy System Model provides an example of a static simulation model (Shalloo *et al.*, 2004). This model has been used to evaluate greenhouse gas emissions along with the performance and economics of Irish dairy production systems (O'Brien *et al.*, 2010). The Sustainable

and Integrated Management Systems for Dairy Production ($SIMS_{DAIRY}$) provides a static modelling framework that integrates other component models to predict ammonia and greenhouse gas emissions, nitrate leaching, P losses and net farm profit for evaluation of dairy production systems (del Prado and Scholefield, 2008). An online tool, I-FARM, is a database-driven farming system model applicable to a wide range of crop and livestock production systems in the eastern US (van Ouwerkerk, 2010).

Dynamic simulation models follow farm processes through time. As such, this type of simulation can better emulate the dynamic interactions among farm components. There are a few good examples of dynamic simulation models that predict various environmental impacts of farming systems. The Farm Assessment Tool (FASSET) is a farm-scale model developed and applied in Denmark, which simulates crop production systems with or without swine and cattle production (Berntsen *et al.*, 2006). The model evaluates farm production, economics and the environmental impacts of nitrate leaching, ammonia emission and net emission of greenhouse gases. The Great Plains Framework for Agricultural Resource Management (GPFARM) includes a dynamic simulation of crop and beef animal production systems for the Great Plains of the US (Ascough *et al.*, 2010). The model simulates soil, crop, pasture and animal processes to predict production performance and economics. Carbon and N cycling are simulated to include environmental losses of ammonia, nitrous oxide, nitrate leaching, erosion and pesticide transport. DairyWise is a dairy farm model that includes N and P cycling, nitrate leaching, ammonia emissions, greenhouse gas emissions, energy use and a financial farm budget (Schils *et al.*, 2007). The Integrated Farm System Model (IFSM) is a comprehensive simulation model of dairy and beef production systems in the USA, which includes the prediction of a wide range of environmental impacts along with production performance and economics (Rotz *et al.*, 2011b). The IFSM model will be described in more detail and will be the focus of the model evaluation and application sections that follow.

In recent years, LCA has become a popular modelling tool. LCA is really an environmental accounting procedure (Heller and Keoleian, 2011; Kristensen *et al.*, 2011). Much information and data must be collected and assumptions made to represent all aspects of each environmental impact over the full life of the milk, meat or other products produced (Pelletier *et al.*, 2010). This includes not only the gaseous emissions and runoff losses that occur directly from the system, but also those occurring during the production of resources used including machinery, fertilizers, fuel, electricity, and purchased feed and animals (Rotz *et al.*, 2010). A full life cycle must include transportation, processing and marketing through to consumption or waste by the consumer. In practice, the LCA of animal production systems has often concluded when the product leaves the farm. This partial LCA represents a cradle to farm-gate evaluation. Economics are normally not included in LCA studies, limiting a full understanding of the impact of a management change to the production system.

Different modelling techniques may be combined in a given software tool. For example, an optimization-type farm model was integrated with a LCA for the determination of sustainable milk production systems in Switzerland (Zimmermann *et al.*, 2011). A partial LCA has also been incorporated in IFSM to evaluate the farm-gate carbon footprint of the milk or meat produced (Rotz *et al.*, 2010). By combining LCA with mathematical optimization or simulation, much of the information required to produce the LCA can be obtained or derived from within other model components. This data input greatly reduces the labour for data collection and the number of assumptions required to develop the LCA. Uncertainties associated with extrapolation or interpolation of location- and time-specific data from existing reported measurements can be large. Thus, using a process-based simulation model that uses known input data to derive estimates of hard-to-measure quantities can reduce accumulated input error for the LCA.

Model purpose

Three primary purposes for farm model development are research, education and decision support. Research tools such as FASSET and IFSM are widely used to evaluate alternative production strategies in search of more economically and

environmentally sustainable systems (Knudsen *et al.*, 2006; Rotz *et al.*, 2011c). A well-developed and evaluated research model provides new information by representing and exploring the effects of interacting components within the farm system. This type of information cannot be readily obtained through experimental measurement or procedures other than dynamic simulation.

Decision support tools are primarily designed to provide useful information and direction for decision making. They are most successful when they address specific and often reoccurring questions or needs of the producer or those consulted by producers. Their purpose is not to develop new information as much as to provide the appropriate information needed to address a current concern and to facilitate the interpretation of available information. GPFARM (Ascough *et al.*, 2001) and I-FARM (van Ouwerkerk, 2010) are examples of whole-farm models developed primarily for decision support purposes. Adoption of whole-farm models in a decision support role has not been as successful as hoped in their early stages of development (Ascough *et al.*, 2010).

Research models can be used in education. For example, IFSM is used in a number of undergraduate and graduate programmes in various universities. Decision support tools can also provide effective educational aids. However, because of the vast difference in goals and information needs, it is infeasible for a single model to be satisfactorily used for both research and decision support. For this reason, components of comprehensive research models are sometimes repackaged to meet the needs of education and decision support. For example, FarmN, a decision support tool extracted from the FASSET model, aids in managing farm N flows (Hutchings and Peterson, 2012). Likewise, the Dairy Gas Emission Model (DairyGEM), developed from the animal and manure-handling components of IFSM, provides a simpler educational tool for evaluating and comparing effects of mitigation strategies on gaseous emissions from dairy farms (Rotz *et al.*, 2011a).

Integrated Farm System Model

The IFSM dynamically assesses and compares the environmental and economic sustainabilities of farming systems (Rotz *et al.*, 2011b). Crop production, feed use and the return of manure nutrients back to the land are simulated for many years of weather on a crop, beef or dairy farm (Fig. 10.1). Crop growth and development are predicted daily based upon soil water and N availability, ambient temperature and solar radiation. Simulated tillage, planting, harvest, storage and feeding operations predict resource use, timeliness of operations, crop losses and nutritive quality of feeds produced. Nutrient contents of available feeds and nutrient requirements of the animal groups making up the herd drive feed allocation and animal responses. Manure quantity and nutrient contents are functions of the herd characteristics and feeds consumed.

Nutrient flows through the farm are modelled to predict nutrient accumulation in the soil and loss to the environment (Rotz *et al.*, 2011b). Environmental impacts include ammonia and hydrogen sulfide emissions from manure sources, soil denitrification and nitrate leaching losses, sediment erosion, and soluble and sediment-bound P runoff. Carbon dioxide, methane and nitrous oxide emissions are tracked for crop, animal and manure sinks and sources to predict net greenhouse gas emission. Secondary emissions that occurred during the production of resources used on the farm, such as purchased feed and animals, fuel, electricity, machinery, fertilizer and pesticides, are included in a cradle to farm gate LCA of the carbon footprint of the milk, meat or feed produced. Whole-farm mass balances of N, P, potassium (K) and carbon are determined as the sum of all imports in feed, fertilizer, deposition and crop fixation minus the exports in milk, excess feed, animals, manure and losses leaving the farm.

Simulated performance is used to determine production costs, incomes and economic return for each year of weather. A whole-farm budget includes fixed and variable production costs (Rotz *et al.*, 2011b). All major production costs are subtracted from the total income received for milk, animal and excess feed sales to determine a net return to management. Each farm simulation, conducted over a 25-year sample of recent historical weather data, results in a distribution of annual predictions that provides an assessment of risk due to varying weather. By comparing simulation results, differences among production systems are determined including annual resource use, production efficiency, environmental impact, production costs and farm profit.

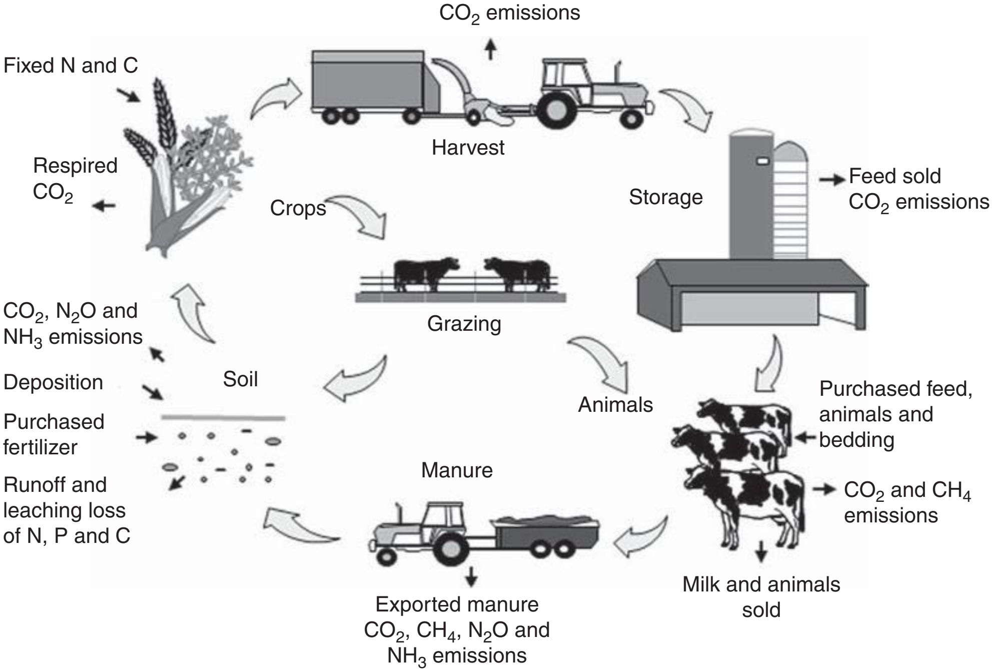

Fig. 10.1. The Integrated Farm System Model simulates all major farm processes, tracking the flow of resources and nutrients to quantify the performance, environmental impact and economics of the production system.

Model Evaluation

Farm models, like all models that answer questions about the natural world, are developed to represent the physical, biological, chemical and/or financial processes of a complex, unbounded and highly dynamic system. Evaluating how well the model represents the real system is always an important aspect of model development and application. This evaluation process is often referred to as model validation. However, the term 'validation' must then be defined as the ability of the model to perform satisfactorily within its intended domain and use (e.g. Refsgaard, 2000). Such a definition does not lend itself to formal, statistical comparisons of model predicted and measured data or to a consistent measure for comparison across system representations or study sites. The many parameters and interacting processes along with the biological uncertainty of those processes make this type of validation infeasible. Additionally, statistical validation requires large amounts of diverse data to quantify all major aspects of the farm. Such data are not available due to the labour required and the cost for measurement and recording at this scale.

Although a formal validation of a farm model is not possible, model evaluation is still important and necessary (Oreskes, 1998; Webster and McKay, 2003). For farm-scale models, there are three levels of evaluation needed. They are model verification, component evaluation and farm-level evaluation. By satisfactorily completing each of these phases of evaluation, the model developer and the user can develop confidence in the ability of the model to represent actual farm production systems. When the model adequately represents the actual system, sensitivity analysis provides valuable feedback on the environmental and production effects of changes to the management system. Similarly, model uncertainty provides a measure of risk to the environmental and production aspects of the farm as a result of unknown or uncontrollable changes in the natural system.

Model verification

Model evaluation must always begin with verification. Verification is the process of confirming that the model is functioning properly, i.e. that there are no errors in the implementation of the model. A model is verified by stepping through each line or section of code within a model to make sure that it is mathematically and logically correct. Many minor and easily corrected coding errors, that would otherwise go unnoticed, can be uncovered with this close evaluation. If left uncorrected, these errors can greatly affect model performance and the use of computer processing and memory resources. With most software development platforms used today, this type of verification is easily done using debugging options. This level of verification requires considerable time though, and it may be overlooked in the desire to complete the model. This verification process is critical. If the model is not functioning properly at this level, other forms of model evaluation and application are irrelevant.

Component evaluation

The next step in model evaluation is the evaluation of individual model components. A complex model such as that representing a farm system consists of many components. These components can include complex system models themselves representing crop growth and development, crop harvest and storage, animal feed intake and performance, and manure-handling systems. As component models such as these are developed and integrated into the larger system model, they must be shown to represent their component properly, both independently and within the full model. For example, as existing crop models were added to the IFSM, they were evaluated to determine how well they represented crop production on farms. Although these crop models were developed and evaluated by others, further evaluation was done to compare predicted crop yields with reported yields from farms over several years of weather to determine not only how well the model predicted farm yields but also the variation in yield from year to year (Rotz *et al.*, 2002a). Other examples of this level of evaluation of IFSM components include the ensiling of feeds (Buckmaster *et al.*, 1989), suitable days for field operations (Rotz and Harrigan, 2005) and animal production (Rotz *et al.*, 2005). This level of evaluation may include statistical comparisons, but often these are more qualitative comparisons of how well model predictions represent the actual farm components. The constraint to performing a more formal validation is normally the lack of well-defined and comprehensive measured data.

Sometimes model evaluation involves newly developed and well-defined relationships for incorporation into the larger model. Examples from IFSM include hay drying and ammonia volatilization. To fulfil a need for simulating the field curing of hay, an empirical model based upon scientific understanding was developed to predict drying rate as a function of soil and weather conditions (Rotz and Chen, 1985). The large amount of field-drying data collected allowed a statistical comparison of measured and predicted drying rates, and hay moisture contents through time. After refining a set of theoretical relationships for predicting ammonia volatilization, the new model was shown to predict measured ammonia emission rates more accurately then other previous models (Montes *et al.*, 2009).

Farm-scale evaluation

Evaluation at the whole farm level is necessary to assure proper representation of the real-world farm system. For this level of evaluation, more general data, often readily obtained from operating farms, are most appropriate for comparison. Such data include crop yields, feeds produced, feeds bought and sold to maintain the herd and the animal products produced and sold from the farm. If a model has been properly verified and evaluated at the component level, an agreement of predicted and actual data at the farm scale can further support an accurate representation of the farm system. Farm-scale evaluation has been used to support the use of the IFSM for evaluating P accumulation on dairy farms in a sensitive watershed in New York (Rotz *et al.*, 2002b), organic dairy production in Pennsylvania (Rotz *et al.*, 2007) and confinement fed and grazing based dairy production in Georgia (Belflower *et al.*, 2012).

When more detailed farm-level data are available, a more comprehensive evaluation is possible. An example is in the evaluation of the nutrient cycling components of IFSM (Rotz *et al.*, 2006). For an experimental farm in the Netherlands called De Marke, comprehensive data were available that tracked the flows, transformations, losses and whole farm balances of N and P. By simulating this farm and comparing predicted and actual values for each aspect of the nutrient cycles, a more complete evaluation of the farm model was made. With this support, the model was then used to study the effect of mitigation strategies on farm performance, environmental impact and economics (Rotz *et al.*, 2006).

For some aspects of the farm, actual data may not be available or quantifiable. For this situation, the model may be best evaluated by comparing with other models and analyses. An example is the carbon footprint of milk production. This type of footprint cannot be measured. By comparing the carbon footprints determined through a number of LCAs with those predicted by IFSM using similar assumptions as those of the original analyses, the accuracy of this new component in the farm model was supported (Rotz *et al.*, 2010).

Sensitivity analysis

Sensitivity analysis determines how farm scale predictions are affected by changes in input parameters or specific functions used within the model. This is done by varying important parameters a set amount like 10% and quantifying how this change affects important output results. The inputs and outputs studied are defined by the application of the model and vary with the problem being addressed. Sensitivity is often quantified as a sensitivity index. This index is the ratio of the per cent change in a given output over the per cent change in the input. An index near or greater than 1 indicates that an output is very sensitive to the given change in the model parameter or function. An index near zero indicates a very low sensitivity.

Model sensitivity is useful in two ways. First, it indicates the amount of error that occurred in the analysis if there was error in the assumed parameter (i.e. parameter uncertainty) or in the function of the model (i.e. structural uncertainty). Therefore, the model developer and user must be as accurate as possible on their assumptions related to very sensitive parameters. The second use is an indication of what can be done to improve system performance. If the result of an analysis is very sensitive to a certain parameter or function and those components have been thoroughly verified, then greater improvements to the farm can be made by strategic or tactical changes that affect that part of the system. Farm production system evaluations with the IFSM often include a sensitivity analysis. Examples are the evaluation of high moisture hay preservation (Rotz *et al.*, 1992), greenhouse gas emissions (Chianese *et al.*, 2009), manure application strategies (Rotz *et al.*, 2011c), organic dairy production (Rotz *et al.*, 2007) and automatic milking systems (Rotz *et al.*, 2003).

Model uncertainty

There are a number of definitions, categorizations and frameworks for incorporating components of uncertainty and variation inherent in complex models (e.g. Refsgaard, 2000; de Rocquigny, 2010). Through model verification and evaluation, the model is confirmed to represent the natural system adequately, and through sensitivity analysis, the impacts to the system caused by intended management changes are determined. Thus, the remaining uncertainty or variation in the modelled output is due to changes in factors that may be known but uncontrollable, such as the weather; beyond the scope of the model; at a much finer or coarser level of detail than appropriate for the model; or simply unknown. For example, spatial variation in soil properties and plant health, individual preferences within a herd, and temporal variations in weather all impact upon crop yield, animal production and nutrient losses to the environment.

With the increased use of models in policy making, uncertainty analysis is becoming more desirable and even necessary, given the uncertain nature of models and the systems they represent. Following the procedure of the IPCC (2006a), the uncertainty of the whole farm emission is the square root of the sum of the squares of the uncertainty of each individual component. The difficulty becomes that of defining the uncertainty of individual parameters or

functions. With large amounts of data for comparison, an uncertainty can be statistically determined. For most farm components though, these data do not exist. An alternative is to define uncertainties through expert opinion. Expert opinion takes advantage of any scientific measurements and knowledge related to a given variable and provides the invaluable and immeasurable contribution of the synthesis of specific and related knowledge by professional experts in the field to determine expected value ranges within the context of the defined boundaries. The IPCC emission assessment methodologies (IPCC, 2006b) and the Chesapeake Bay Model (EPA, 2010) both use expert opinion to determine the uncertainty in their prediction of farm management effects on air and water quality. For IFSM, the uncertainty of predicted greenhouse gas emissions has been estimated through expert opinion (Chianese *et al.*, 2009).

For farm systems, a crucial and easily modelled form of uncertainty is that due to the influence of weather. Daily and annual predictions of nearly all aspects of the farm are controlled by weather. By simulating farm performance over multiple years, the IFSM predicts variation in environmental impacts and farm economics as influenced by weather (Rotz *et al.*, 2011b). This enables an evaluation of annual risks of particular management practices with regard to profit, environmental losses and other model outputs. Large standard deviations in model predictions highlight those that are highly sensitive to annual weather patterns. Thus, on a year-to-year basis, managing the farm to achieve a certain value for these parameters is high risk. But in the long term, the effects average out reducing risk.

Production System Evaluations

To demonstrate the effects of management on farm profitability, the environment and the interactions between air and water quality, several farm simulations were done with the IFSM. These simulations consist of a few examples of animal, manure handling and crop management in dairy and beef farm systems. The simulations include predictions for major pathways of nutrient losses, whole farm balances of the major nutrients and the environmental footprints of reactive N loss, energy use and carbon emission. Within each management group, strategic changes are compared to illustrate the resulting environmental effects along with the producer's costs and profit.

These comparisons of simulated farming systems are provided just as an illustration of the use of the model and the interacting effects of farm management on the environment. Only brief descriptions of the simulated representative farms are presented. In-depth detail on all model assumptions cannot be provided and the results should not be viewed as conclusive comparisons of the simulated management options. The results are specific to the farms simulated and changes in farm parameters can influence the results obtained. These results do provide good examples of the trade-offs between various environmental impacts and the costs and benefits to the producer.

Animal management

To demonstrate the effects of animal management, four management options were simulated on a rather large dairy farm in Idaho. The dairy herd included 3000 lactating and non-lactating cows with 2600 replacement heifers raised on the farm. For the base farm, the annual replacement rate of the lactating cows was 40% with an annual milk production of 10,000 kg per cow. Animals were housed in free stall barns with access to open lots. Manure was flushed from the barns and stored in a lined earthen lagoon for application to cropland in the spring and fall. Manure solids were separated with 50% of the solids exported to remove a portion of the nutrients from the farm. Cropland consisted of 800 ha of lucerne and 400 ha of maize on a clay loam soil in a gently sloping terrain. A conservation tillage system was used to establish crops, and all cropland was irrigated to improve production. Except for a small amount of side-dressed N fertilizer, all crop nutrient requirements were met through manure nutrients produced on the farm. The lucerne hay and maize silage produced provided two-thirds of the forage requirement of the herd with the remainder of the forage and all concentrate feed purchased. Animals were fed total mixed rations with protein and P overfed by 10% to assure that requirements were met.

The first management change was to formulate diets precisely to meet each animal group's requirements for degradable and non-degradable proteins and P (Rotz *et al.*, 2011b). The reduction in protein reduced the N excreted in manure, which reduced ammonia emissions from animal housing, manure storage and following field application by about 8% (Table 10.1). Losses through nitrate leaching and denitrification were also reduced by about 10%. Overall, this provided an 8% reduction in the total reactive N loss from the production system. The reduction in P fed did not have much effect on P runoff loss, but the accumulation of excess P in the farm soil was reduced by 45%. These feeding changes provided a small 3% reduction in greenhouse gas emissions and carbon footprint. More precise feeding was of benefit to the producer. The reduction in purchased feed cost increased annual farm profitability by US$35 per cow.

These simulation results illustrate that more precise feeding to meet animal nutrient requirements benefits both the environment and the producer. This assumes that production is not negatively affected by this change. Producers often overfeed nutrients to reduce their risk of loss of production. With accurate measurement

Table 10.1. Simulated annual environmental and economic impacts of animal management on a 3000-cow dairy farm in Idaho.

	Base farm[a]	Precision feeding[b]	Reduced replacement rate[c]	Improved production[d]
Nutrient loss and balance (kg ha^{-1})				
Nitrogen lost by volatilization	110.0	101.1	102.1	112.4
Nitrogen lost by leaching	10.5	9.4	9.5	11.1
Nitrogen lost by denitrification	132.6	119.2	120.1	140.1
Phosphorus loss by runoff	0.3	0.3	0.3	0.3
Soil phosphorus accumulation	13.9	7.7	12.4	16.7
Soil potassium accumulation	43.1	37.6	22.5	50.5
Sediment erosion loss	287	287	287	287
Ammonia emission (kg NH_3 per cow)				
Animal housing	27.7	25.9	25.4	28.2
Manure storage	16.4	14.8	15.8	16.6
Field application	9.4	8.4	8.5	9.9
Total	53.5	49.2	49.7	54.7
Greenhouse gas emission (kg CO_2e per cow)				
Feed production	1011	974	958	1051
Animal	4335	4240	3887	4493
Manure storage and handling	2602	2519	2410	2715
Total	7948	7733	7255	8259
Reactive nitrogen footprint (g N kg^{-1} milk)	4.74	4.35	4.45	4.46
Fossil energy footprint (MJ kg^{-1} milk)	3.03	3.01	2.86	2.96
Carbon footprint (kg CO_2e kg^{-1} milk)	1.03	1.00	0.97	0.99
Production costs ($ per cow)				
Net feed production and use	1877	1843	1744	2000
Manure handling	83	82	78	86
Animal management, milking, etc.	1087	1087	1085	1092
Total	3047	3012	2907	3178
Income from milk and animal sales (US$ per cow)	3711	3711	3646	4050
Net return to management (US$ per cow)	664	699	739	872

[a]3000 cows plus 2600 replacement heifers on 1200 ha of cropland (800 ha of lucerne and 400 ha maize) annually producing 10,000 kg of milk (3.5% fat) per cow. [b]Protein and phosphorus are fed more precisely, reducing requirements by 10%. [c]Annual replacement rate of the lactating herd is reduced from 40% to 30%. [d]Through genetic improvement, milk production is increased from 10,000 to 11,000 kg per cow.

or knowledge of the nutrient contents of available feeds, more precise formulation and feeding of animal diets is a good option for improving animal management.

With the goal of maximizing milk production, the replacement rate of the milking herd has increased in recent years (CDIC, 2011). This requires the production of more replacement heifers to maintain the herd, which requires more feed production and increased manure handling. A long-term management option may be to improve the longevity of the milking animals through genetic improvement or better animal care. This option was modelled by reducing the replacement rate of the milking herd to 30% and the number of replacement heifers raised on the farm to 2000.

This change in animal management had a similar effect on N losses as the reduction in protein feeding just described (Table 10.1). Ammonia emission was reduced by about 7%, nitrate leaching by 10% and the reactive N footprint by 6%. Maintaining fewer replacement animals had little effect on P runoff loss, but the accumulation of excess P and K were reduced. Net greenhouse gas emission and carbon footprint were also reduced about 9%. Increasing longevity in lactation was also beneficial to the producer. Farm income was reduced slightly due to fewer cull cows sold, but this was offset by reduced production costs increasing annual farm profitability by US$75 per cow. Therefore, improved animal management to reduce the replacement rate of the herd is beneficial to both the environment and the producer when it is done without sacrificing milk production.

Through improved animal genetics and feeding management, milk production per animal has steadily improved for many years and this trend continues (NASS, 2011). Producing more milk with the same or fewer animals can provide both environmental and economic benefits. This is illustrated by increasing the milk production level of the herd to 11,000 kg per cow (Table 10.1). To produce the additional milk, feed and nutrient intakes are increased, which increases nutrient excretion in manure (Rotz *et al.*, 2011b). This creates a small increase in ammonia emission with a 6% increase in nitrate leaching. Considering the greater amount of milk produced, the total reactive N loss per unit of milk is reduced by 6%. An environmental concern though, is that the long-term increase in soil P and K is increased, which could lead to greater nutrient runoff losses in the future. Greenhouse gas emissions are also increased about 4% but the carbon footprint (emission per unit of milk produced) decreased 4%. Increased milk production is of great benefit to the producer. Increased feed and manure-handling costs were offset by increased milk sales increasing annual farm profit by over US$200 per cow.

Manure handling

The effects of four manure-handling options are illustrated using a smaller dairy farm in central Pennsylvania. The dairy herd consisted of 100 Holstein cows and 80 replacement heifers with an annual replacement rate of 35% and milk production of 9500 kg per cow. All animals were fed total mixed rations to meet their nutrient requirements. Free stall barns were used to house animals with manure removed daily by scraping. Crops included 40 ha of lucerne, 50 ha of maize and 10 ha of oats produced on gently sloping clay loam soil. Small amounts of N, phosphate and potash fertilizers were used to meet crop nutrient requirements along with all manure nutrients produced on the farm. Lucerne and maize silage stored in bunker silos provided all of the forage needed to feed the herd. The remaining maize and oats were harvested, stored in a tower silo and fed as high moisture grain. With this cropping strategy, 87% of the total feed requirement was produced on the farm. Straw from the oat crop was also used for bedding.

For the first manure-handling option, daily hauling and field application of manure was used (Table 10.2). Manure was scraped from the free stall barns, loaded on a spreader and applied to cropland each day. The manure was broadcast on the field surface without incorporation into the soil. As a second option, a concrete tank was used to store manure for field application in the spring and autumn, and manure was incorporated into the soil by a tillage operation within 2 days of application. Compared with the daily hauling practice, use of long-term manure storage increased ammonia emission by 32% but decreased nitrate leaching and denitrification losses by 14% (Table 10.2). Overall, the total loss of reactive N was increased 9%. Energy use was

reduced a small amount through fewer round trips hauling manure to the field. Greenhouse gas emissions and the carbon footprint were increased about 30%, primarily due to the increased emissions from the manure storage. The cost of constructing the manure storage increased production costs, which reduced the producer's annual profit by US$38 per cow.

As a third option, a sealed cover was used on the manure storage where any biogas produced was burned, releasing carbon dioxide to the atmosphere (Rotz *et al.*, 2011b). The cover reduced ammonia emission from the storage structure, but much of the retained ammonia was later released following field application (Table 10.2). Nitrate leaching, erosion and P runoff losses were similar to those occurring with the open storage. Greenhouse gas emissions and the carbon footprint were similar to those occurring with the daily haul system, but the reactive N footprint was increased 6%. The added cost of covering the manure storage reduced the producer's profit an additional US$44 per cow.

Table 10.2. Simulated annual environmental and economic impacts of manure management on a 100-cow dairy farm in central Pennsylvania.

	Daily hauling of manure[a]	6 month slurry storage[b]	Covered slurry storage[c]	Covered storage plus injection[d]
Nutrient loss and balance (kg ha^{-1})				
Nitrogen lost by volatilization	43.6	57.5	54.6	20.7
Nitrogen lost by leaching	46.2	39.8	41.2	57.5
Nitrogen lost by denitrification	26.8	23.1	23.9	33.1
Phosphorus loss by runoff	1.9	1.9	1.9	1.6
Soil phosphorus accumulation	0.0	0.0	0.0	0.0
Soil potassium accumulation	0.0	0.0	0.0	0.0
Sediment erosion loss	3444	3387	3387	3384
Ammonia emission (kg NH_3 per cow)				
Animal housing	19.0	18.9	18.9	19.1
Manure storage	0.0	9.5	1.5	1.5
Field	34.0	41.5	46.0	4.6
Total	53.0	69.9	66.4	25.1
Greenhouse gas emission (kg CO_2e per cow)				
Feed production	2340	2219	2240	2207
Animal	4328	4322	4323	4332
Manure storage and handling	153	3002	180	172
Total	6821	9543	6743	6711
Reactive nitrogen footprint (g N kg^{-1} milk)	9.23	10.07	9.80	8.06
Fossil energy footprint (MJ kg^{-1} milk)	3.17	3.12	3.12	3.15
Carbon footprint (kg CO_2e kg^{-1} milk)	0.88	1.15	0.87	0.87
Production costs (US$ per cow)				
Net feed production and use	1474	1484	1483	1476
Manure handling	306	331	373	412
Animal management, milking, etc.	1363	1366	1369	1369
Total	3143	3181	3225	3257
Income from milk and animal sales (US$ per cow)	3740	3740	3740	3740
Net return to management (US$ per cow)	597	559	515	483

[a]100 cows plus 80 replacement heifers on 100 ha of cropland (40 ha of lucerne, 50 ha maize and 10 ha oats with straw for bedding) annually producing 9500 kg of milk (3.5% fat) per cow. Animals are housed in free stall barns where manure is collected and applied to cropland each day. [b]Same as a except manure is collected in a bottom-loaded storage tank and applied to cropland in the spring and autumn through broadcast application with tillage incorporation within 2 days. [c]Same as b with a sealed cover on the manure storage with a flare to burn accumulated biogas. [d]Same as c with manure applied through injection below the soil surface.

As a fourth manure-handling option, equipment was added to the manure applicator to inject the manure below the soil surface. Subsurface injection can reduce gaseous emissions and the surface runoff of nutrients (Rotz *et al.*, 2011c). Use of this practice on this farm reduced field emission of ammonia by 97%, which reduced the total farm emission by about 50% (Table 10.2). Nitrate leaching and denitrification losses were increased though, so the total loss of reactive N was reduced about 20%. Placing the manure below the surface reduced P runoff by 16%. Greenhouse gas emissions and the carbon footprint were not affected by this added operation, but the energy use was increased slightly due to the additional time and power needed to inject the manure. With the cropping strategy used on this farm (abundance of N available through legume fixation by the lucerne), there was little economic benefit received through more efficient N use. The increased cost of the injection equipment and the added fuel and labour required reduced farm profit an additional US$32 per cow.

As illustrated through these simulations, changes in manure-handling practices can be made to reduce nutrient, particularly N, loss to the environment. Reduced ammonia emissions can be beneficial in improving air quality, but if changes in the cropping strategy are not made to improve the utilization of the retained N, greater losses through nitrate leaching and denitrification will occur. Changes in manure-handling practices often come at a cost to the producer, reducing farm profit. Therefore, there is little incentive for producers to make these changes unless they are reimbursed through increased prices or other societal payments.

Crop management

To illustrate the effects of crop management, a beef farm was simulated in southern Pennsylvania. The herd consisted of 150 Angus cows and their offspring including 40 replacement heifers and 100 growing cattle. The land base was 130 ha of loam soil with a sloping terrain. Calves were weaned at 7 months of age and maintained as stocker cattle for 5 months. During these periods, animals were fed pasture or other forage to maintain appropriate growth rates. For the final 4 months, growing cattle were finished on a high grain diet. Throughout the year, cows and replacement heifers were maintained on pasture and harvested forage with most of the manure deposited directly on pasture land. During the finishing period, cattle were maintained in a bedded barn with that manure applied to cropland in the spring and autumn.

For the first cropping option, the land base consisted of 80 ha of perennial grassland and 50 ha of maize. The maize land was tilled with a mouldboard plough followed by a series of conventional tillage operations to prepare the soil for planting. About two-thirds of the maize was harvested and fed as maize silage. Controlled grazing was not used, which affected both yield and nutritive content of the pasture forage. Animals maintained on pasture obtained about 60% of their feed from grazing with the remainder from maize silage. With this tillage practice on this sloping terrain, the erosion of sediment and loss of sediment-bound P were very high (Table 10.3).

As a second option, a no-till strategy was used for establishing maize, i.e. all maize was planted using a zone till planter with no other tillage operations. All other aspects of the farm remained the same except for increased use of pesticides for weed and insect control. The reduced tillage provided a large decrease in erosion and P runoff with very minor effects on N losses and greenhouse gas emissions (Table 10.3). With fewer tillage operations, energy use in the production system was reduced 3% and this provided a very small reduction in the carbon footprint. The reduced cost for tillage operations was partially offset by increased pesticide costs leaving an increase in annual farm profit of US$37 per cow.

For a third option, a winter cover crop was used following maize silage harvest. Annual rye was established soon after harvest in late summer to maintain ground cover over the winter. In the spring, the crop was killed prior to spring planting of maize. During the spring and summer, degradation of the crop residue returned nutrients back to the soil for use by the growing maize crop. The additional soil cover reduced the erosion of sediment an additional 30%, and the cover crop uptake and turnover of N reduced nitrate leaching by 10% (Table 10.3). Gaseous emissions were not affected by this change in

Table 10.3. Simulated annual environmental and economic impacts of crop management on a 150-cow beef farm in southern Pennsylvania.

	Conventional tilled maize[a]	No-till maize[b]	No-till maize with cover crop[c]	All perennial grassland[d]
Nutrient loss and balance (kg ha^{-1})				
Nitrogen lost by volatilization	43.9	44.1	44.3	48.9
Nitrogen lost by leaching	36.5	37.0	33.3	15.5
Nitrogen lost by denitrification	17.2	16.7	16.3	7.2
Phosphorus loss by runoff	1.9	0.6	0.5	0.2
Soil phosphorus accumulation	0.0	0.0	0.0	0.0
Soil potassium accumulation	8.8	8.6	9.1	0.0
Sediment erosion loss	6624	1056	732	227
Ammonia emission (kg NH_3 per cow)				
Animal housing	9.0	9.0	9.1	10.1
Manure storage	3.1	3.1	3.1	3.1
Field application	6.6	6.9	6.9	9.1
Pasture	27.5	27.5	27.5	29.2
Total	46.2	46.5	46.6	51.5
Greenhouse gas emission (kg CO_2e per cow)				
Feed production	745	728	735	741
Animal	4265	4262	4279	4671
Manure storage and handling	442	442	446	451
Total	5452	5432	5460	5863
Reactive nitrogen footprint (g N kg^{-1} BW[e])	141	142	136	114
Fossil energy footprint (MJ kg^{-1} BW)	29.6	28.7	36.1	23.3
Carbon footprint (kg CO_2e kg^{-1} BW)	13.1	13.0	13.5	13.6
Production costs (US$ per cow)				
Net feed production and use	650	615	667	474
Manure handling	46	46	46	46
Animal management, etc.	196	196	196	196
Total	892	857	909	716
Income from animal sales (US$ per cow)	1033	1033	1035	1033
Net return to management (US$ per cow)	141	176	126	317

[a]150 cows with 40 replacement heifers on 80 ha of grassland and 50 ha of maize annually producing 100 finished cattle. When not on pasture, animals are housed in a bedded barn where manure is collected and applied to maize land semi-annually with mouldboard plough incorporation. [b]Same as a except that no-till crop establishment is used without manure incorporation. [c]Same as b with a rye cover crop established after maize silage harvest (on 30 ha of maize land) and killed in the spring for recycling of nutrients. [d]The entire farm (130 ha) is used as rotationally grazed perennial pasture with excess pasture harvested for winter feeding. [e]BW, body weight.

cropping strategy, but energy use was increased 25% through the added planting operation. Increased costs for seed, pesticides and the added operations reduced the producer's profit by US$50 per cow.

As a final option, the entire farm was converted to perennial grassland where a rotational grazing strategy was used to supply all of the forage needed to maintain the herd. Grain used to finish cattle was purchased and imported to the farm. More controlled grazing practices provided higher nutrient contents and better utilization of pasture forage. Excess pasture growth in the spring and summer was harvested and stored as round bale silage or hay. This forage was used to maintain the herd during the winter months and other periods when adequate pasture was not available.

Using more pasture forage in animal diets increased their protein intake, which led to

greater N excretion and thus greater ammonia emission (Table 10.3). Better N utilization was maintained throughout the year with perennial grassland, which reduced nitrate leaching and denitrification losses by over 50%. More complete year-round ground cover also reduced sediment erosion and runoff loss of P, and greater use of grass allowed a whole farm balance of the major nutrients. The elimination of maize harvest operations reduced the energy footprint by about 25%. Greater fibre intake by the animals increased enteric methane emission by about 10%. This increase in methane emission along with the reduction in energy use provided a small increase in the carbon footprint of beef production. This change in crop management was of benefit to the producer reducing annual feed costs and increasing farm profit by over US$140 per cow.

Many different cropping strategies can be used to provide feed in livestock production, and any given strategy can have both positive and negative environmental impacts relative to other options. This diversity makes this type of evaluation specific to the farm or production system being evaluated, and general conclusions cannot be made. Reduced tillage operations and the use of cover crops will normally be beneficial for water quality, but they may or may not be of benefit to air quality or of economic benefit to the producer. Greater use of perennial grassland will normally benefit most aspects of water quality, but there may be trade-offs for air quality and carbon footprint. The economic costs and benefits of grassland and controlled grazing are greatly influenced by the size and type of livestock operation.

Beyond the Farm

As just illustrated, farms can create a number of environmental impacts to both air and water quality. There are often trade-offs when mitigating these impacts such that reducing one air pollutant may increase another air or water pollutant. The pollutant of most concern will vary among regions, so some type of prioritization among pollutants may be required when selecting mitigation strategies. For example, in many regions in the eastern USA surface runoff of nutrients is of greatest concern, so greater amounts of gaseous emission may be tolerated if this can lead to better water quality. In the drier climate of the western US, air emissions are often of greater concern than surface runoff. Ultimately though, strategies are needed that benefit both air and water quality while maintaining profitable livestock production systems.

The farm is only a piece of the whole system of pollution and the management required to reduce pollution. As nutrients leave the farm, they affect groundwater, surface water and air beyond the farm. Because groundwater moves relatively slowly, groundwater pollution is typically noticed locally where drinking water from wells can be affected within several kilometres of the farm. Surface water affects a larger area, normally defined as the watershed, consisting of streams and water bodies used for recreational purposes as well as drinking and fishing. Gaseous emissions such as ammonia affect the airshed of the region. Airsheds are not well defined as they are influenced by wind speed and direction, and the topography of the region. Pollutants entering the air may travel great distances and transform in the atmosphere before they are deposited back to the earth. Greenhouse gases remain in the atmosphere for many years raising atmospheric concentrations, which are believed to be affecting climate change. These effects reach far beyond the region, affecting global atmospheric conditions.

Watershed evaluation

Watershed boundaries can be accurately defined to track the flow of water toward streams, rivers and larger water bodies. Watershed models are often used to evaluate the impact of changes in agricultural management at the watershed outlet by simulating nutrient losses from agricultural land and the transport of those nutrients through the watershed. Two commonly used watershed-scale water quality models are the Annualized Agricultural Non-Point Source Pollution Model (AnnAGNPS) (Bingner and Theurer, 2012) and the Soil and Water Assessment Tool (SWAT) (USDA-ARS, 2012). Processes occur during stream flow, which transform and filter nutrients, so the quantity and type of nutrients leaving the farm are not

the same as that reaching the major waterways or watershed outlet (Sharpley *et al.*, 2009). Some work has been done to link farm and watershed models (Ghebremichael *et al*; 2013), but more of this work is needed to ultimately define the impact of farms on water quality at the watershed outlet and beyond.

Non-point-source pollution from agriculture has continued to be one of the major causes of water quality degradation in streams and lakes of the USA. Agricultural non-point source pollution controls are commonly addressed through various federal and state conservation programmes providing financial and technical assistance and through voluntary use of cost-share best management practices by landowners. The successes of non-point source pollution control efforts depend upon proper identification, targeting and remediation of critical source areas (CSAs) of pollution. CSAs within a watershed contribute proportionally more pollutants to the outlet than other areas. CSAs for surface runoff represent an overlap of high pollution source areas with areas prone to generate high volumes of runoff and erosion. Their identification through ground-truthing, geographic information systems (GIS) and watershed-scale modelling have helped guide P index development and the placement of best management practices on farms. CSAs for groundwater can play a significant role in aquifer pollution when the region is highly fractured, shrink-swell, tiled or karst; however, these areas are difficult to define and delineate.

Airshed evaluation

An airshed is the atmospheric area where the majority of a given emission is deposited back to a land region. Airsheds are often many times larger than the watershed of the same region (Fig. 10.2). For example, nitrous oxide emissions from an airshed 6.5 times larger than the Chesapeake Bay watershed, contribute 76% of the oxidized-N deposition reaching that watershed (Paerl *et al.*, 2002). These emissions represent one-third of the nitrous oxide emissions in eastern North America.

Although size variations among airsheds are smaller than variations among watersheds in the eastern US (Paerl *et al.*, 2002), CSAs for air pollution are very difficult to delineate as transport factors are continually changing. Locating a particular contributor, or source, of non-point source emission within the watershed can be difficult. Confirming that the majority of emissions from that source redeposit directly to the water body, the majority of the time, relies on complex models. Models are being developed and used to predict the transport, transformation and deposition of N and other constituents (e.g. Wang and Chen, 2012), but little has yet been done to link farm scale prediction of emissions to airshed scale transport and deposition models. As one example, Del Grosso *et al.* (2010) used a biogeochemical model called DAYCENT to simulate nitrous oxide emissions from cropland soils across the USA. They found spatial variability of the emissions to depend mainly on differences in N inputs via fertilizer and manure application, whereas temporal variability was driven by N mineralization caused by the weather. To understand the ultimate impact of farm emissions such as ammonia, better integration of farm and airshed scale models is needed.

Future needs in the integration of air and water quality issues

Addressing air and water quality in an integrated manner is becoming increasingly vital to address the needs of a growing population along with pollution weakening the natural resiliency of terrestrial ecosystems, oceans and the atmosphere. Increases in human population and their requirements for food, fibre and fuel are increasing nutrient cycle imbalances (Paerl *et al.*, 2002). Corresponding increases in water use and changes in land use are likewise impacting the hydrological cycle, decreasing streamflow and precipitation (Wang *et al.*, 2009). Atmospheric carbon dioxide is projected to increase through the 21st century (WMO, 2010). Likewise, ozone depletion and climatic warming will continue, particularly if methane and nitrous oxide are not strictly controlled (WMO, 2010). Carbon dioxide deposition into the oceans is causing an increase in acidification that is detrimental to marine life (Hardt and Safina, 2010). EPA is working to better integrate local ecosystem and watershed

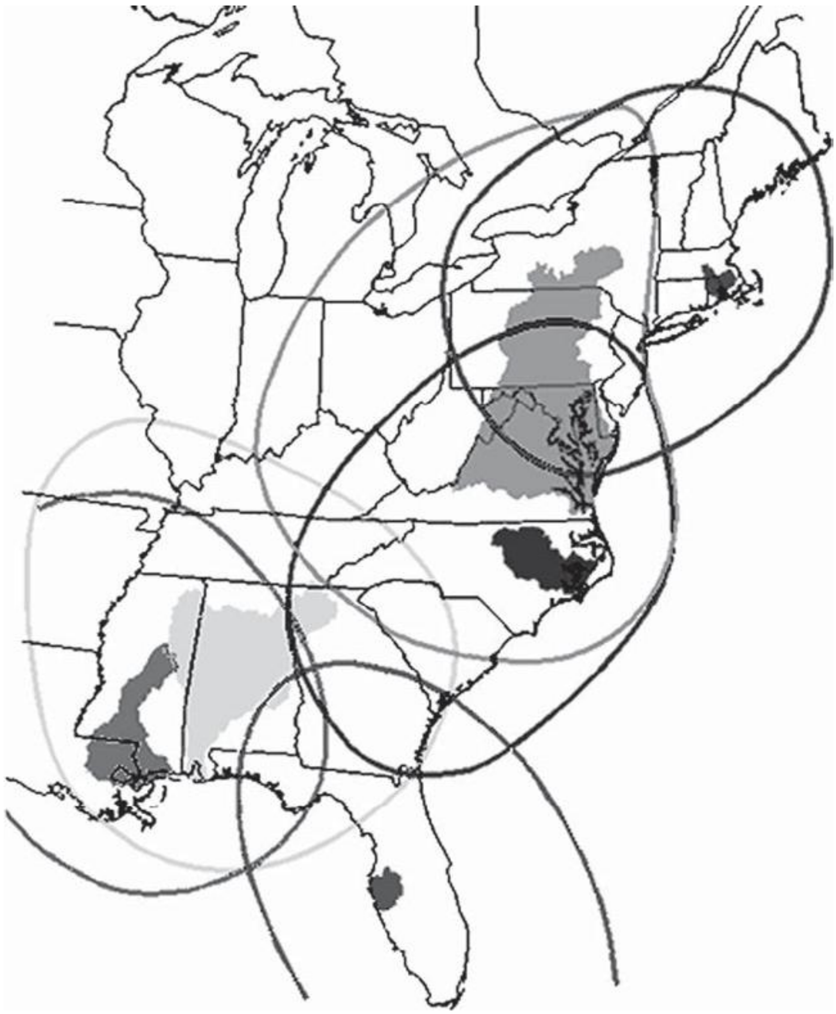

Fig. 10.2. Watersheds and oxidized nitrogen airsheds for five major eastern US waterbodies (from EPA, 2011).

models with regional airshed models to predict pollution effects through space and time (EPA, 2011).

Changes in climate are anticipated to create greater extremes in the weather, which will likely lead to larger 'flushing' events. The concentration in the air and water during these events will often exceed the assimilation capacity of the environment. In the case of water quality, this would result in increasing hypoxic zones. In the case of air quality, the diffusion of high pollutant concentrations will increase airshed boundaries, resulting in deposition of pollutants from one watershed into multiple watersheds. Over time, as the hydrologic cycle rotates pollutants attached to water vapour through water movement, evaporation and precipitation, pollutants will spread more and more widely throughout the world.

These developing concerns for our environment support the need for greater integration of air and water quality issues. This type of integration can only be addressed through complex process-level modelling. For livestock agriculture, this must begin at the farm. Producers ultimately control their production systems. They are normally good stewards of their land and

desire to reduce their negative impacts and increase their benefits to the environment. However, producer decisions are necessarily driven by the profitability of production. Farm-level mitigation practices will be adopted to reduce nutrient losses to the environment, but this must be done while maintaining a profitable livestock industry.

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11 The Environmental Sustainability of Food Production

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Introduction

Sustainability is defined by the US Environmental Protection Agency (US EPA, 2010b) as 'meeting society's present needs without compromising the ability of future generations to meet their own needs'. This definition is a relatively simple example of one of the myriad that are present in the lexicon of food industry stakeholders, indeed, sustainability is multi-faceted and represents many differing concepts and practices to different organizations or individuals. However, there appears to be a growing consensus that sustainability, regardless of the precise definition, may be the most important issue faced by the food production industry. Until recently, sustainability referred simply to economic viability and the resilience of an organization or industry to survive market shifts and volatility. As sustainability becomes increasingly important for a growing proportion of consumers, the definition has shifted to a 'triple-bottom-line approach', which encompasses three factors: economic viability, environmental stewardship and social responsibility. Short- or long-term sustainability relies on a balance between these three factors. If one factor is out of balance, e.g. if a production practice is economically viable and reduces environmental impact yet is unacceptable to the consumer, the system is out of balance and can only be resolved by improving social responsibility or removing the practice from the system. For example, the use of gestation crates in swine production is considered to have economic benefits to the producer (Krieter, 2002) and potentially to reduce environmental impact by increasing sow productivity. However, a number of processors and retailers have recently removed this management practice from their supply chain in response to public perceptions of sub-optimal animal welfare under this management system.

Following the earlier definition from US EPA (2010b), environmental sustainability may be summarized as 'producing more, using fewer resources'. On a global basis, the seeming dichotomy between ensuring food security and reducing resource use for food production is a significant issue for governments and policy makers who are conscious not only of the proportion of their national population that is currently food-insecure, but also of the Food and Agriculture Organization of the United Nations (FAO)'s prediction that the global population will increase to over 9.5 billion people in the year 2050 (FAO, 2009). The extent of population growth varies amongst regions, with the greatest increases in both population growth and per

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capita income predicted to occur in developing nations such as Africa, China and India (Tilman *et al.*, 2002). The demand for high-quality animal proteins such as meat, milk and eggs increases concurrently with per capita income, thus the total food production may have to increase by 70% by 2050 in order to supply the global population with a nutritionally adequate diet (FAO, 2009).

Greenhouse gas (GHG) emissions are often the focus of environmental sustainability issues, with a significant amount of media and research attention being paid to reducing anthropogenic carbon emissions. However, competition for resources (including water, land and energy) between food production and other human activities may be a more significant issue in the event of considerable population growth, and the global livestock industry will face a significant challenge in producing sufficient animal protein to meet consumer demand, using a finite resource base. This issue is not confined to a future scenario – current concern over dwindling natural resources and climate change leads to debate as to whether the livestock industry should continue to intensify and improve productivity to feed the increasing population, or return to less-productive traditional methods. This chapter will focus on past, current and future prospects for food production sustainability, with a particular focus on productivity and sustainability within ruminant livestock systems.

The Role of Maintenance Nutrient Requirements in Environmental Sustainability

The recent rise of 'ethical consumerism', defined by Singer and Mason (2006) as 'an interest in the way in which food is produced, the practices employed and a concern for low environmental impact, high animal welfare and optimal worker conditions', has the potential to significantly influence the management practices and systems within livestock production. The mantra that 'the consumer is always right' applies equally to livestock production as to any other industry, yet given the amount of misinformation relating to how food is produced circulating within the media and popular press, a danger exists that decisions may be based upon philosophical ideology rather than scientific foundation. The FAO (2006) concluded that intensification of animal production was necessary on a global basis to reduce carbon emissions from livestock systems. None the less, intensive livestock production systems may be at the greatest risk from a social responsibility standpoint, as such systems are often perceived to be environmentally damaging. Although it is widely understood that improving efficiency and productivity reduces expense, resources and waste, the consumer often considers 'efficiency' to have negative connotations when applied to large-scale contemporary food production.

Every animal has a daily nutrient maintenance requirement that must be fulfilled to support vital functions and minimum activities, and is independent of whether or not the animal is actively producing an animal protein product. For example, a lactating cow produces milk, whereas a young calf does not, yet they both have a daily maintenance requirement. The total maintenance requirement of the livestock population may be considered as a 'fixed cost' of animal production. To understand the environmental impact of improved efficiency livestock production better, an economic metaphor can be used in which fixed costs are a proxy for population maintenance nutrient requirements (Capper, 2011b). Consider a bakery producing bread with fixed costs of US$1000 (rent, taxes, etc.) incurred each day, regardless of productivity. If the factory produces 10,000 loaves per day, the fixed costs can be divided by the total output (US$1000/10,000 loaves = US$0.10 per loaf) and the bread priced accordingly. If the bakery improves productivity so that 20,000 loaves are manufactured in the same period, efficiency improves and fixed costs are spread over greater output (US$0.05 per loaf). The same concept can be applied to livestock production and is known as the 'dilution of maintenance' effect. Improving productivity such that a greater quantity of milk, meat or eggs is produced in a set time reduces the total maintenance cost per unit of food produced. This directly affects environmental sustainability as maintenance nutrients may be considered a proxy for resource use (including feed, land, water and fossil fuels) and waste output (e.g. manure and GHG). Improving productivity consequently reduces resource use and waste output per unit of food. This relatively simple goal has been the foundation for

improving environmental sustainability of the US ruminant livestock industries over the past century (Capper *et al.*, 2009; Capper, 2011a). None the less, the mechanisms by which productivity improves, and the management systems or production practices implemented to promote productivity gains, vary significantly between industries and species.

The Dilution of Maintenance Concept within Ruminant Production

Improving yield per animal is the most widely understood productivity measure. If we increase the milk yield of a lactating dairy cow from 22 to 31 kg day^{-1}, the maintenance energy requirement (75 MJ day^{-1}) does not change, but is diluted out over more units of production and thus reduced from 42% to 34% of total daily energy needs. Concomitantly, the energy required per kg of milk is reduced from 8.1 to 7.2 MJ kg^{-1} (Fig. 11.1). As milk yield increases, fewer lactating cows are required to produce a set amount of milk and the number of associated support animals (dry cows, replacement heifers, bulls) in the population decrease, thus the total population maintenance requirement is reduced. The environmental impact of increased milk yield may be exemplified by comparing resource use and GHG emissions from US dairy production in 1944 and 2007 as described by Capper *et al.* (2009).

The US dairy herd peaked at 25.6 million cows in 1944, with an average annual milk yield per cow of 2074 kg and an annual national production of 53.0 billion kg milk (Capper *et al.*, 2009). At this time, dairy herds characteristically contained six cows that were fed a pasture-based diet with occasional supplemental feed. Artificial insemination was in its infancy and neither antibiotics nor supplemental hormones were available for animal use. By contrast, the 2007 US dairy herd contained 9.2 million cows producing 84.2 billion kg milk (average yield 9193 kg year^{-1}). These productivity gains were facilitated by improvements in management, nutrition, genetics and the application of new technologies and are a proof of concept for the dilution of maintenance effect reducing both resource use and GHG emissions through a decrease in dairy population size. Compared with 1944, the 2007 US dairy industry required only 21% of the dairy population to produce a set quantity of milk (Fig. 11.2). Consequently, feedstuff use was reduced by 77%, land use by 90% and water use by 65% per unit of milk. Manure output per unit of milk produced in

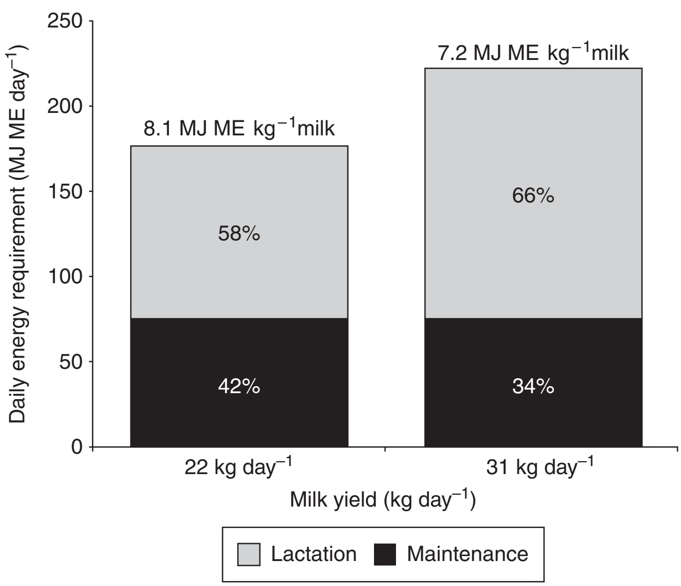

Fig. 11.1. The impact of increasing daily milk yield on the proportion of daily energy used for maintenance versus lactation in dairy cattle – the 'dilution of maintenance' effect.

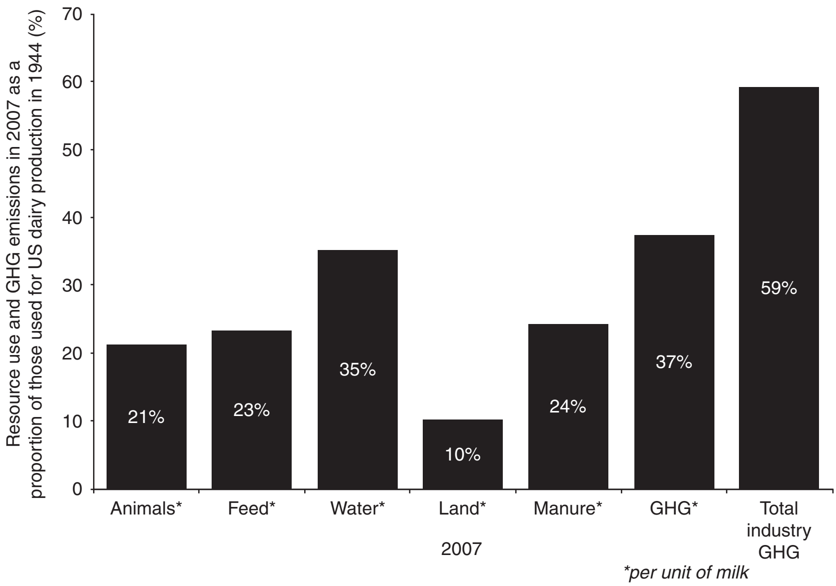

Fig. 11.2. Comparative resource use and greenhouse gas (GHG) emissions for US dairy production in 1944 and 2007.

2007 was 24% of that in 1944 and the total carbon footprint per unit of milk was reduced by 63%. Despite the increase in total milk production between 1944 and 2007, the total carbon footprint for the entire dairy industry was reduced by 41%.

Similar yield trends have been exhibited by the US beef industry, which, as a result of improved genetic selection, ration formulation and growth-enhancing technology use, has increased slaughter weight and thus beef yield per animal over time. Between 1977 and 2007, average beef-carcass yield per animal increased from 274 to 351 kg (USDA, 1978; USDA/NASS, 2008). This allowed beef production in 2007 to be maintained from a population containing 30% fewer animals (cows, calves, heifers, bulls, stocker and feedlot animals) than the equivalent population in 1977 (Capper, 2011a). As per the previous dairy example, this diluted the population maintenance requirement over greater units of production. As maintenance requirements are primarily dictated by bodyweight, the increased slaughter weights in 2007 meant that the average daily maintenance requirement per animal was increased; yet this was overcome by improvements in growth rate, which reduced the proportion of daily energy requirements associated with maintenance from 53% to 45% and the total days from birth to slaughter from 609 to 485 (Fig. 11.3), thus reducing the lifetime maintenance requirement. The combination of improved beef yield and growth rate conferred decreases in feed use, water use and land use per unit of beef of 19%, 12% and 33%, respectively. Manure output per unit of beef was reduced by 18%, with a 16% decrease in the carbon footprint per unit of beef (Capper, 2011a). Anecdotal evidence from the beef processing industry suggests that average slaughter weights have plateaued and that further increases in slaughter weight are undesirable given the current processing infrastructure and consumer portion size demands. Therefore improvements in growth rate may offer a greater opportunity for future beef industry sustainability than further yield increases.

The Role of a Reduced Maintenance Requirement in Improving Environmental Sustainability

Improved productivity has a demonstrable impact on beef's environmental sustainability; however,

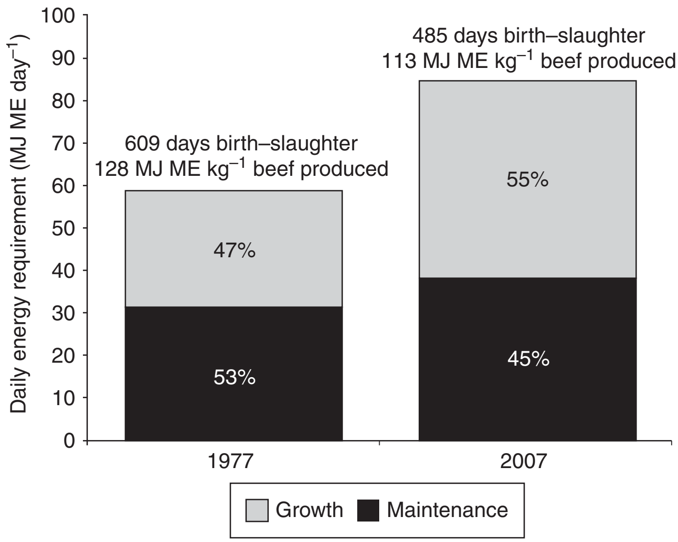

Fig. 11.3. The comparative daily energy requirements for maintenance and growth of beef cattle produced under management systems characteristic of US beef systems in 1977 and 2007.

discussion exists within the beef industry as to whether selection for growth rate and slaughter weight has indirectly reduced the efficiency of the mature cow. Within an efficient cow–calf system, calves should be weaned at approximately half the dam's mature weight, yet this is difficult to achieve when mature cows approach or exceed 635 kg bodyweight given the nutritional limitations of pasture-based diets. Life cycle assessments (LCA) of beef production also indicate that the cow–calf sector contributes the greatest proportion of carbon emissions per unit of beef (Beauchemin *et al*., 2011) and is less susceptible to dietary mitigation of methane emissions due to the extensive nature of the production system. Although Notter *et al*. (1979) reported some advantages of increased mature weight upon cow–calf system efficiency, this author predicts that if calf growth rates and weaning weights can be maintained from cattle with a mature weight of less than 580 kg, environmental sustainability will be improved. The question therefore becomes whether future environmental sustainability may be achieved through further dilutions of maintenance, or whether a reduction in the maintenance requirement of the supporting population through changes in mature weight may play a significant role.

The US dairy industry has made a wholescale shift away from smaller breeds to larger-framed cows over the past century – within the previously discussed historical comparison, the 1944 dairy population contained 54% smaller breeds (Jersey and Guernsey) compared with 90% Holstein cows in 2007 (Capper *et al*., 2009). Genetic selection for increased milk yield has increased the proportion of Holstein cattle within the national dairy herd, yet one criticism often levelled at intensive dairy production systems is that high-producing cows tend to have an increased bodyweight, and thus consume more feed and emit greater quantities of GHG on a daily basis. If daily GHG emissions per animal are the correct metric by which to evaluate environmental impact, cows with a greater mature weight will have increased GHG output compared with their smaller counterparts – for example, the daily GHG output per lactating cow was 13.5 kg CO_2-eq in 1944 compared with 27.8 kg CO_2-eq in 2007 (Capper *et al*., 2009). None the less, expressing results on a 'per head' basis fails to consider milk yield, milk composition and other productivity indices that may have a significant effect upon population size and thus environmental impact. As the population maintenance nutrient requirement has a sizeable impact upon resource use and GHG emissions per unit of dairy product, the impact of reducing population maintenance through changes in mature cow weight warrants further investigation.

Jersey cattle confer two potential breed-specific advantages over the Holstein in terms of

environmental impact. First, they have an increased milk solids concentration (480 g kg^{-1} fat and 370 g kg^{-1} protein compared with 380 kg milk fat kg^{-1} and 310 g protein kg^{-1} for the Holstein) and thus a predicted Cheddar cheese yield of 125 g kg^{-1} milk compared with 101 g kg^{-1} milk (Capper and Cady, 2012). Despite their reduced milk yield (20.9 kg day^{-1} compared with 29.1 kg day^{-1}), predicted daily cheese yield is therefore only slightly less than the Holstein (2.61 versus 2.94 kg). Second, mature US Jersey cattle have an average bodyweight of 454 kg compared with 680 kg for the Holstein, thus individual animals have a smaller maintenance requirement. If equivalent quantities of Cheddar cheese production were produced from Jersey and Holstein cow populations, the assumption that dairy population size can be used as a proxy for environmental impact does not hold true. Although the interaction between milk yield and milk solids concentration meant that the Jersey population required to produce 500,000 Mt of cheese yield contained 9% more animals than the Holstein population, the body mass of the Jersey population was reduced by 26% and the population maintenance energy requirement by 22%. Consequently, water use was reduced by 32%, land use by 12% and GHG emissions by 20% per unit of cheese yield. Within this comparison, the major factors affecting resource use and GHG emissions were milk yield, milk solids content and animal bodyweight, thus although productivity is a key contributor, it is not the sole arbiter of livestock sustainability. The concept of reducing maintenance requirements is also being adopted within both the beef and dairy industries through selecting animals with a low residual feed intake (RFI), i.e. those animal that require less feed to support maintenance and production than they would be predicted to consume (Herd and Arthur, 2008; Crozier and ZoBell, 2010). If future genetic indices are developed to the extent that producers are able to select effectively for low RFIs, this would be predicted to have a significant impact on environmental sustainability through improved efficiency.

Environmental Sustainability of Organic and Pasture-based Systems

The social acceptability of organic or pasture-based systems is generally improved compared with conventional livestock production. Demand for organic and 'natural' foods is increasing in developed countries where malnutrition is more often associated with obesity than undernourishment, and consumers have sufficient income to demand greater food choice. In the USA, organic food commands a small portion (3.7%) of total market share (Organic Trade Association, 2010) with the greatest market shares being seen in the fruit and vegetable (12%) compared with dairy (6%) or beef (2.5%) sectors (Clause, 2010; Organic Trade Association, 2011). Recent data show that almost 95% of US consumers buy food according to economic, nutritional and taste aspects, with only 4% seeking food according to their specific lifestyle choices (e.g. vegetarian, organic or local), and yet a majority of consumers will occasionally buy organic foods (Simmons, 2011). A survey by Raab and Grobe (2005) reported that consumers associated organic foods with positive attributes including 'chemical-free', 'healthier/more nutritious', 'clean/pure' and 'earth-friendly'. It is clear that the intensification of US livestock production over the past century has considerably reduced both resource use and GHG emissions per unit of animal protein. None the less, a small yet vocal proportion of the population advocate for pasture-based or organic systems (Pollan, 2007; Salatin, 2007; Gumpert, 2009), citing differences in the nutritional quality or environmental impact of animal proteins produced in extensive systems.

Pelletier *et al.* (2010) reported that GHG emissions per unit of beef were greater in pasture-finished systems than in feedlot systems. This result seems intuitively incorrect; a conventional system that finishes animals on maize-based diets grown with significant fertilizer inputs, transports both feed and animals across the country, and houses animals in confinement appears to have intrinsically lower environmental sustainability than a grass-finishing system. None the less, from a biological viewpoint, the results are easy to explain. Growth rates are considerably less in animals finished on grass and it is difficult to achieve heavier slaughter weights, therefore grass-finished cattle are usually slaughtered at around 486 kg at 679 days of age, compared with 569 kg at 444 days of age in a conventional system (Capper, 2012). As a consequence of the reduced slaughter weight,

4.5 total animals (slaughtered animals plus the supporting population required to produce calves for rearing) are required to produce 363 kg of hot carcass weight beef in a grass-finished system compared with 2.6 total animals in a conventional system. When combined with the increased time required for animals to grow to slaughter weight, this increases the carbon footprint per unit of grass-finished beef by 67.5%. The increased land required for grass-finished production renders whole-scale conversion of the US beef production system to grass-finished production practically impossible. However, if we assume it would somehow be achievable and that beef production was maintained at 11.8 billion kg as in 2010 (USDA/NASS, 2011), the increase in carbon emissions would be equal to adding 25.2 million cars to the road on an annual basis (Capper, 2012).

Proponents of pasture-based beef finishing systems may argue that increased GHG emissions from grass-finished cattle are compensated for by the quantity of carbon sequestered by pastureland. However, pasture does not sequester carbon indefinitely, nor does it occur at a constant rate. Over time, soil carbon concentrations reach an equilibrium point, beyond which no further sequestration occurs unless land is subjected to significant management change (Post and Kwon, 2000; Schlesinger, 2000). The present body of knowledge indicates that the degree to which carbon may be sequestered by crop or pastureland is infinitely variable between systems and is dependent on a myriad of factors including land use change, tillage, organic matter input, soil type and crop/pasture species. Reliable data on carbon sequestration under a range of environmental conditions and global regions are notably lacking from environmental literature and this is one area where future research would pay dividends in bridging the current knowledge gap. Although the majority of US beef animals are finished within feedlot systems, pasture and forage-based diets are fed to these animals for half to two-thirds of their life, and diets for the supporting beef herd (cows, heifers and bulls) are predominantly based on forage. Any potential effect of carbon sequestration in mitigating GHG could hence only be attributed to the finishing period. In the aforementioned comparison between conventional and grass-finishing, sequestration would need to exceed 1.3 t of carbon per ha annually (Capper, 2012). This is a considerable target, given that Bruce *et al.* (1999) suggest that the potential for carbon sequestration in well-managed pastureland is 200 kg ha^{-1}, whereas Conant *et al.* (2001) report 540 kg ha^{-1}. Furthermore, although carbon emissions from the finishing population would be mitigated by this degree of sequestration, the greater supporting population (cows, calves, heifers and bulls) conferred by reduced slaughter weights in the grass-finished system would also need to be accounted for.

In the event that productivity in grass-based finishing beef systems could be improved to that commonly exhibited by maize-based finishing, sequestration would still need to occur in order to compensate for the propensity for pasture-based diets to increase ruminal methanogenesis and thus enteric GHG emissions (Johnson and Johnson, 1995; Pinares-Patiño *et al.*, 2009). Methane production from enteric fermentation is not a new phenomenon within the scientific community, yet the link between climate change and livestock production is a relatively recent notion. Consumers therefore often perceive that modern livestock production causes climate change, whereas historical livestock populations were far more environmentally friendly. To put this historical supposition into context, Capper (2011b) noted that the 60 million American bison that roamed the US plains until mass extinction in 1880 had total GHG emissions (based on enteric methane production and GHG emissions from manure) approximately double the carbon emissions from the US dairy industry in 2007 (Fig. 11.4).

Results from studies comparing resource use and carbon emissions from organic or pasture-based dairy systems vary considerably according to methodology and region (de Boer, 2003). Cederberg and Mattsson (2000) reported reduced total GHG emissions per functional unit of milk from organic versus conventional dairy farms despite a reduction in milk yields in the organic system. A computerized simulation of European dairy production by Olesen *et al.* (2006) demonstrated a tendency for organic farms to have higher GHG emissions per unit of milk than conventional farms, yet in this example, milk yields were assumed equivalent between systems. Given the considerable contribution of enteric methane to total GHG emissions per unit

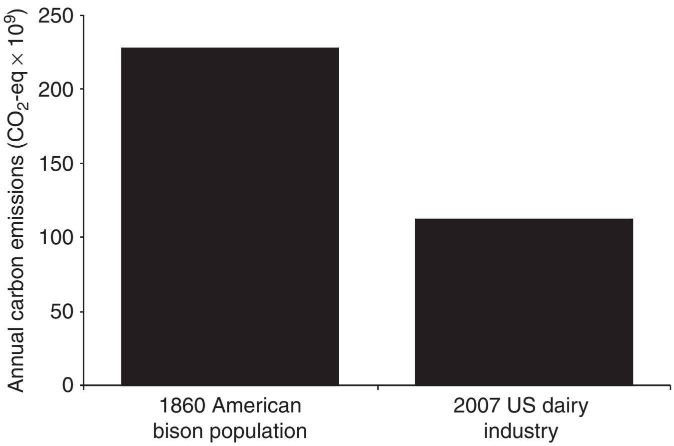

Fig. 11.4. Comparative annual carbon footprints of the 1860 American bison population and 2007 US dairy industry, adapted from Capper (2011b). The carbon footprint for American bison is based on CH_4 and N_2O emissions resulting from forage dry matter intakes for age-appropriate bodyweights and population dynamics, emission factors are from US EPA (2007).

of pasture-based dairy, it is logical to suggest that such systems may only gain a significant environmental advantage over conventional dairying when they support milk production without negatively affecting milk yield. However, USDA (2007) data revealed a 26% decrease in milk yield per cow in organic compared with conventional systems, and peer-reviewed papers comparing organic and conventional production cite decreases in milk yield ranging from 14% to 40% (Zwald *et al.*, 2004; Sato *et al.*, 2005; Rotz *et al.*, 2007). Assuming that dairy breed and bodyweight remain static, a reduction in milk yield means that the dairy population size must increase in order to maintain total fluid milk production. Projecting out to the year 2040 when the US population is predicted to plateau at 340 million people, supplying the entire population with their USDA-recommended 0.71 l of low-fat milk (or its equivalent) per day through organic rather than conventional production practices would require 3.5 million additional animals to be added to the national herd (Capper *et al.*, 2008). Land use would concurrently increase by 3.1 million ha (a 30% increase) and the carbon footprint of organic dairy production would be 13% greater than that of conventional production. It may be suggested that the rise of ethical consumerism (Singer and Mason, 2006) and associated preference for specific production system or management practices is a direct consequence of the greater incomes, food availability and consumer choice enjoyed by consumers within developed countries compared with less developed regions of the world. Indeed, an African official from the FAO was quoted within a newspaper report of a conference dedicated to genetically modified food as stating that: 'Organic farming is practiced by 800 million poor people in the world because they can't afford pesticides and fertilizers – and it's not working' (McNeil, 2000).

Environmental Sustainability of Monogastric Production Systems

Compared with ruminant production systems, swine and poultry industries are generally considered less environmentally threatening with regards to resource use and GHG emissions. Estimates of the carbon footprint of monogastric animal protein production range from 2.8 to 4.5 kg CO_2 kg^{-1} pork (Strid Eriksson *et al.*, 2005; Vergé *et al.*, 2009; de Vries and de Boer, 2010) and 1.9 to 2.9 kg CO_2 kg^{-1} chicken (Katajajuuri, 2008; Pelletier, 2008; Cederberg *et al.*, 2009). None the less, given the increase in poultry and swine consumption predicted to occur over the next

40 years (Rischkowsky and Pilling, 2007), further efficiency improvements may be necessary within these industries to reduce future environmental impact. Vertical integration and consolidation within both industries has considerably improved productivity over the past 50 years. According to historical USDA data, between 1963 and 2009, average US swine carcass weight increased by 27 kg, from 65 to 92 kg (USDA, 2012). This allowed total carcass weight (slaughtered animals × average carcass weight) to increase from 5.4 to 10.5 billion kg (a 92% increase) while slaughter numbers only increased by 44% (35 million animals). Despite the increase in slaughter numbers, the US swine breeding population declined from approximately 9.1 million head to approximately 5.9 million head, as a function of both increased litter size and a greater number of farrowings per year. In contrast to the beef system, where multiple offspring (twins or triplets) are generally perceived to be undesirable, improved reproductive performance in terms of increased litter size is therefore a significant factor in swine productivity. Indeed, Thoma *et al.* (2011) demonstrated a positive impact of increased litter size on carbon emissions from swine production, and Vergé *et al.* (2009) noted reductions in GHG emission intensity resulting from higher birth rates in the Canadian pork industry between 1991 and 2001.

Average chicken slaughter weight also increased from 1.61 to 2.54 kg between 1963 and 2009, facilitating a 594% increase in chicken production (3.17 to 29.4 billion kg) with only a 3.4-fold increase in slaughter numbers (1.96 to 8.66 billion head; USDA, 2012). Poultry growth rates and feed efficiency also improved considerably over the past 60 years, reducing the time from hatching to slaughter from 90 days to less than 40 days (Konarzewski *et al.*, 2000). Evidence from feeding studies involving heritage-style chicken breeds suggests that although nutrition and management have played a significant role, the majority of this improvement has occurred through genetic gain (Havenstein *et al.*, 2003; Schmidt *et al.*, 2009). Although environmental analysis of gains made in the US pork or poultry industry over time have yet to be executed, results of the aforementioned historical beef comparison suggest that the increases in monogastric carcass weight and growth rates, and reductions in supporting population sizes would be expected to have mitigated the environmental impact per unit of pork or poultry over time.

The Role of By-product Feeds in Animal Agriculture and Livestock's Competition for Human Food

A question is often posed as to whether livestock systems can achieve further productivity gains in future without significant genetic or technological intervention, both of which may be unpalatable to the consumer. This is further exacerbated by consumer concern over the use of crops for animal feed that could instead be used as human foodstuffs (Gill *et al.*, 2009). Given the feed efficacies and growth rates already exhibited within monogastric production systems, there may be less opportunity to improve these metrics than in ruminant livestock. However, monogastric diets are primarily based on maize and soybean, thus the use of by-products from the human feed and fibre industries, which have a considerably smaller carbon footprint (as the majority of carbon emissions can be attributed to food or fibre), may be a potential avenue to further mitigate GHG emissions from monogastric animal production. Ruminant systems provide for the conversion of human-indigestible plant material (forages, pastureland and by-product feeds) into high-quality animal protein for human consumption. The majority of land used for grazing ruminants is not suitable for growing crops for human consumption, indeed data from the USDA's Economic Research Service (Lubowski *et al.*, 2006) indicate that only 8% of US grazed land is sufficiently productive to be classified as cropland pasture, therefore using pastureland to support, for example, the cow–calf sector of beef production provides an opportunity to feed the human population without competing for grain-based food resources. This is discussed at length by Wilkinson (2011), who redefined conventional measures of feed efficiency (e.g. 7.8 kg feed per kg of gain for feedlot-finished beef compared with 3.6 kg feed per kg of gain for pork) to account for human-edible energy or protein feed inputs compared with the

human-edible energy or protein output from livestock production systems. Within these metrics, grass-finished beef had a favourable human-edible feed efficiency ratio whether expressed in terms of energy (1.9 MJ/MJ human-edible energy in animal product) or protein (0.92 kg kg^{-1} human-edible protein in animal product) when compared with pork (2.6 kg kg^{-1} human-edible protein) or poultry (2.1 kg kg^{-1} human-edible protein). It is therefore clear that feed efficiency *per se*, is not an adequate measure by which to compare differing animal proteins from a resource use basis.

The Environmental Sustainability of Omnivorous Human Diets

The suggestion that crops used for animal feed could be better used to supply human nutritional needs naturally leads to the question as to whether animal agriculture is a necessary component of the human diet. A considerable number of reports have recently claimed that consumers could considerably reduce their carbon footprint by forgoing animal protein on one or more days per week (Fairlie, 2010; Millward and Garnett, 2010; Environmental Working Group, 2011). The related 'Meatless Mondays' campaign appears to originate from a paper published by Weber and Matthews (2008) in which the authors state that 'Shifting less than one day per week's worth of calories from red meat and dairy products to chicken, fish, eggs, or a vegetable-based diet achieves more GHG reduction than buying all locally sourced food.' The lack of a 'control' treatment against which to compare the removal of red meat and dairy products from the diet renders this comparison practically meaningless, none the less, it has been adopted by various vegetarian and vegan groups as proof that meat consumption is environmentally unfriendly. Citizens of most developed nations could arguably consume less red meat and dairy products without negatively impacting their health status, none the less, it is somewhat disingenuous to suggest that this dietary change would have a major impact on national or global GHG emissions. For example, the US population is generally considered to have the greatest regional red meat consumption per capita at 49.4 kg (CME Group, 2011); yet GHG emissions attributable to red meat and dairy production are equal to 3.05% of the national total (US EPA, 2010a). If every US inhabitant removed red meat and dairy products from their diet, the reduction in US GHG emissions would be equal to 0.44%. Any attempt to reduce GHG may be considered laudable, none the less, a 0.44% reduction (assuming that this concept was adopted by the entire population) would make very little difference to total emissions, especially since it is not expressed in the context of other human activities, for which we have at best a tenuous grasp of the potential environmental impact. For example, a significant reduction in meat consumption would also necessitate development of non-animal-based replacements for products such as manure, leather, adhesives and pharmaceuticals that are derived from livestock production.

Fairlie (2010) published an elegant calculation comparing the food-supply potential of land given the dietary preferences of the UK population. Results demonstrated that under the constraints of food production for a conventional, omnivorous population whereby agriculture is based on a mixture of livestock and chemical fertilizer inputs, 1.0 ha of arable land plus 1.5 ha of pastureland would feed 14 people. By contrast, 1.0 ha of arable land would supply 20 people with a vegan diet, yet fertilizer inputs would have to be entirely based on chemical inputs, which would considerably increase fertilizer and fossil fuel requirements per unit of food. If the agricultural system shifted to one based on vegan permaculture (without regard for the limitations of using human sewage to fertilize crops intended for human consumption), 1 ha of land would feed eight people. In addition to the lower potential food supply, the increased dependence on arable land within vegan systems and lack of opportunities to use pastureland, thus reducing potential improvements in biodiversity and soil structure from proper grazing management would also be expected to impact environmental sustainability. The debate as to which human diet provides the 'optimal' use of resources is likely to continue for some time, and is unlikely to ever gain resolution given that optimal resource use is region- and population-specific.

The authors of a recent Environmental Working Group report (2011) should be commended

in that the results are presented on the basis of maintaining dietary protein supply from animal versus plant sources, although the authors demonstrate a limited understanding of livestock production systems, which in some cases lead to underlying assumptions for their model that are entirely unfeasible. For example, the methodology indicates that the reproductive productivity of the US sheep system (farm flocks average 1.5–2.5 lambs per ewe per year whereas range flocks average 1.0–1.5 lambs per ewe) is not accounted for compared with the US average for beef of 0.87% of cows producing a live calf (USDA, 2003, 2009). Approximately twice as many beef cows are required to produce the same number of offspring as a set number of ewes; therefore, even considering the fact that lambs produce slightly less meat per carcass on a percentage basis, this throws doubt on the Environmental Working Group's claim that the GHG emissions associated with lamb are approximately 44% greater than those of beef. Nutritionally, however, it is a fallacy to suggest that protein quality does not differ between various animal and plant sources, especially when specific amino acids are required for human growth and development (Bauman and Capper, 2011). Just as a recent study by Drewnowski (2010) evaluated the economic cost per unit of dietary food energy, evaluation of the environmental impact of dietary protein sources should be related to protein quality in order make dietary recommendations without negatively affecting human health.

The need to assess resource use and GHG emissions based upon the nutrient density of individual food products is of increasing importance to processors and retailers as product differentiation continues to be used as a mechanism to gain market share. Some European retail grocery chains have adopted labelling schemes displaying the 'carbon footprint' per unit of food, and such schemes may be adopted in the USA in future. However, this is a particular concern when comparing, for example, fluid milk to cheese. Given that it takes between 8 and 10 kg of milk to make 1 kg of cheese, a unit weight of cheese would be labelled with a carbon footprint approximately ten times that of the same unit weight of milk. This might lead the consumer to discriminate against products that have a larger carbon footprint, regardless of nutritional value. Smedman *et al.* (2010) reported that milk had the most favourable ratio of nutrient content to GHG emissions when compared with orange juice, soy and oat beverages and alcoholic drinks; none the less, the challenge is to communicate these results to consumers to whom animal products are often regarded as either nutritionally unfavourable or environmentally unsustainable.

Opportunities for Future Global Environmental Sustainability

In 2006, the FAO released the oft-quoted report 'Livestock's Long Shadow', which concluded that livestock production is responsible for 18% of global anthropogenic GHG emissions. Despite its adoption by the majority of media and activist groups as scientific evidence for the principal role of livestock in causing climate change, the FAO report was not without its detractors and Pitesky *et al.* (2009) produced a detailed paper outlining the flaws within the report. The most notable issue related to the 18% statistic, which was derived from comparing carbon emissions from a highly detailed and inclusive LCA of global livestock production, to the carbon emissions from the fuel combustion phase of the global transport sector. As the FAO later admitted, differences in the methodology between predicted carbon emissions from livestock production and transport rendered the comparison invalid. The exact proportion of global carbon emissions produced by livestock production has yet to be quantified although it is suggested to be somewhat less than the original 18% estimate. Although the FAO report was regarded as damaging by many within the livestock industry, it fulfilled two vital roles with respect to environmental sustainability – the magnitude and shock-value of the 18% figure ensured that climate change became a priority for industry groups, and carbon emissions from all livestock sectors came under scrutiny.

As previously discussed, the environmental sustainability of livestock systems is profoundly affected by system productivity, yet the FAO's global average includes a wide range of regional efficiencies. The environmental effects of regional productivity variations are exemplified

by the results of a second FAO (2010) report that focused on modelling global and regional GHG emissions from dairy production using LCA. As dairy system intensity declines and the average milk yield shifts from approximately 9000 kg per cow for North America to ~250 kg per cow for Sub-Saharan Africa, the carbon footprint increases from 1.3 to 7.6 kg CO_2-eq kg^{-1} milk (Fig. 11.5). The productivity and efficiency gains made by the US livestock industries over the past century have conferred significant improvements in environmental sustainability, yet it is crucial to note that true sustainability can only be achieved by making the best use of the resources available within each region; thus it is dangerous to consider any specific system as a utopic ideal that should be replicated elsewhere.

Regional livestock system sustainability should therefore not be limited to the environmental impact of the system, but must also consider the economic and social implications. For example, the data in Fig. 11.5 could provoke the conclusion that all regions should adopt North American and Western European-style production systems in order to reduce the carbon footprint of dairying, or that production should be focused in these areas and be discouraged in less productive regions such as Sub-Saharan Africa and South Asia. However, the significant social (both status and nutritional) and economic value of dairying in less-developed regions must not be underestimated. The challenge for global livestock production is to optimize sustainability within each region, specifically developing regions that will bear the brunt of increased population growth within the next 40 years, rather than prescribing the best 'one-size-fits-all' system.

Conclusions

The livestock industry faces a clear challenge in producing sufficient animal protein to supply

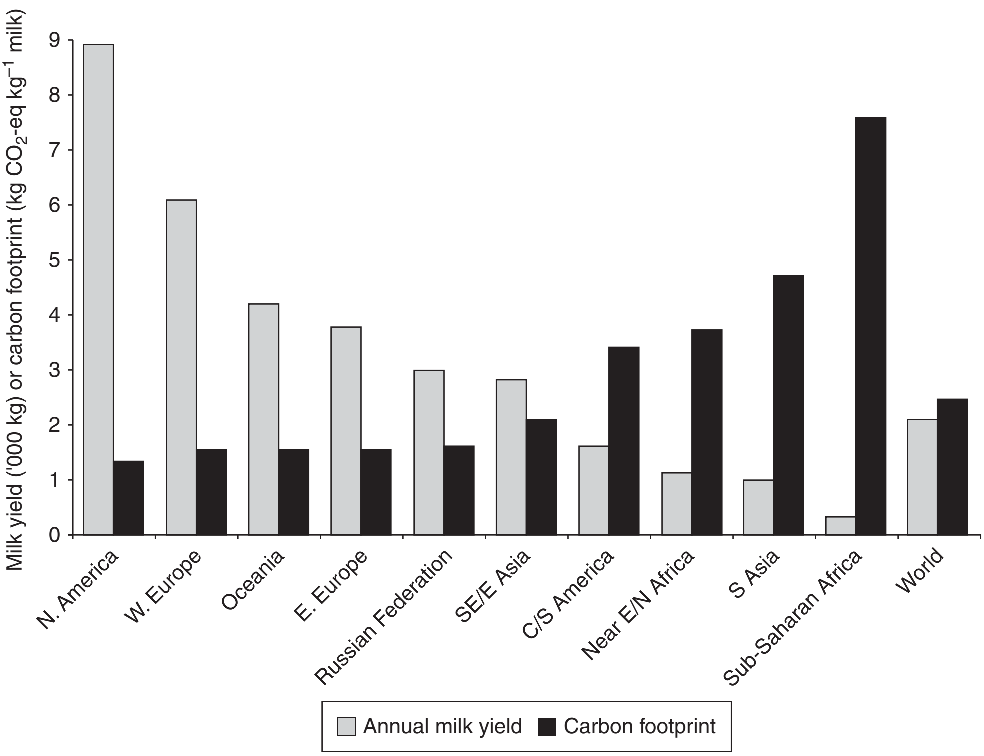

Fig. 11.5. Average annual milk yield and carbon footprint per kg of milk for selected global regions, adapted from Capper (2011b).

the needs of the growing global population, while continuing the tradition of environmental sustainability. The beauty of consumer choice within the developed world lies in the fact that there is a market for every production system, intensive or extensive, large-scale or small-scale, conventional or alternative, with or without technology use, providing it continues to adapt to the economic, environmental and social issues that together confer sustainability. In order to fulfil the dietary requirements and desires of the growing population it is essential to improve productivity within all systems without demonizing or idolatrizing particular systems or practices. In regions where food is readily available, consumers are afforded the luxury of making dietary choices, yet many developing regions exist where the simple need for food negates such concerns. The global livestock industry must therefore identify system-specific and region-specific sustainable practices to ensure that dietary needs are met and environmental sustainability improved without prescription of a one-size-fits-all solution.

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12 Economic Sustainability in Animal Agriculture

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Introduction

Food animal producers are in the middle of a dynamic economic sector that is undergoing significant changes, which will continue for decades. Livestock producers are between suppliers of feed (and other inputs) and animal processors in the flow of products within the 'meat product' sector, and all of the industries in this sector are shifting in structure. Structural shifts cause changes in both the conduct and performance of an industry, thus many changes will occur in food animal agriculture in the near future. Probably the most important shift is the ongoing concentration of animal processing firms. The structural shifts caused by the concentration of buyers for livestock (i.e. animal processing firms) are enabling the exercise of market power, which hastens the shifts in structure, thus the shifts are inevitable. The ultimate result of these changes is that integrated vertical supply chains will grow more common and stronger over the next decade. This could have serious implications for US livestock producers and, possibly, consumers.

As agribusiness firms, agricultural producers and policy makers have become aware of the expanded scope of change, a new debate is focusing on an old issue: 'sustainability'. Both natural and social scientists are now focusing much attention on this issue, yet those efforts are rarely coordinated because the underlying problems and components are not yet well understood because they reflect complex systems and phenomena. As a result, progress toward sustainability has been very limited (Toman, 1994).

The general objective of this chapter is to contribute to the understanding of both the economic issues behind the changing structure of livestock industries and the likely implications of those changes for the sustainability of food animal agriculture as it will exist in the future. In this effort, two specific objectives are pursued. First, the limited amount of data available at this early stage in the trend toward increased use of market power are summarized; pointing to an explanation for what is driving that trend. The second specific objective is to present the implications of the trends in industry structural changes. Based on a discussion of previous research results, preliminary inferences are drawn on the future of animal agriculture in America – its potential for success as well as the fundamental limitations for livestock producers as they pursue economic sustainability.

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What is Sustainable Agriculture?

The term 'sustainability' itself has not been defined universally by natural and social scientists, thus each group has a unique perspective of what it means and, therefore, how it should be studied (Common and Perrings, 1992). In a natural science perspective, 'sustainability' generally focuses research on identifying the balance and renewability of natural systems. In economics, however, 'sustainability' is viewed as a temporal constraint on decision-making.

Economic studies of 'sustainability' focus on the 'fairness' of outcomes between periods (Foy, 1990; Toman, 1994). For example, the simplest of these economic models evaluate outcomes between the present and future generations. One outcome of economic assessments of sustainability has been the development of the 'sustainability criterion'. The criterion suggests that, at a minimum, future generations should be left no worse off than current generations (Tietenberg, 2003). Combining the perspectives of natural and social scientists, and especially economists, means that, together, these groups must identify and/or create and help the industry select agricultural production systems that satisfy present and future needs.

The definition of the term 'sustainable agriculture' is still evolving. A computer search for the definition generates at least one statement for each of the natural and social sciences trying to address the challenge. Even on the campuses of the numerous universities with 'centres' or 'institutes' or 'programmes' focusing on sustainable agriculture, there are multiple definitions being used by different groups or departments. Unfortunately, scientists tend to focus narrowly within their discipline and sustainability requires a broad, multidisciplinary view.

The US Congress tried to come to the rescue in 1990. They defined sustainable agriculture in the 1990 Farm Bill. Given that federal funding is a substantial source for agricultural research at present, it would be wise for all natural and social scientists to become familiar with the specifics of that definition. Under that law (US Congress, 1990), 'the term sustainable agriculture means an integrated system of plant and animal production practices having a site-specific application that will, over the long term:

- satisfy human food and fiber needs;
- enhance environmental quality and the natural resource base upon which the agricultural economy depends;
- make the most efficient use of non-renewable resources and on-farm resources and integrate, where appropriate, natural biological cycles and controls;
- sustain the economic viability of farm operations; and
- enhance the quality of life for farmers and society as a whole.'

Two things should catch the eye about this definition of sustainable agriculture. The first theme in the definition is that the five bulleted points outline tasks that require both natural and social scientists to accomplish. The first bulleted point requires social scientists to identify the 'needs' and natural scientists to create the systems necessary to produce the output sustainably. The second bullet point is nearly all natural science, except that environmental and resource economists might contribute in the discussions of regulatory and market frameworks. The third point involves economists to identify 'efficient' uses and natural scientists to fulfil the remainder of the tasks. The fourth point is all economics. The final point is again a combination of natural and social science tasks. Thus, no scientific discipline has a monopoly on finding a path to the goal – it must be a multidisciplinary effort.

The second theme is that 'economic viability' (or sustainability) is a subset of the definition. Economic sustainability is key in identifying what *will* happen: it is a necessary, but not sufficient, condition for achieving the goal of a sustainable agriculture. This means a significant contribution to be made by economists is to evaluate the profitability of agricultural operations, not only in monetary amounts, but more importantly in terms of return on investment so as to facilitate comparisons between alternative systems developed by natural scientists. Part of this contribution will be analysing the markets in which profitability is determined. The structure of markets for agricultural outputs is critical in the search for a sustainable agriculture, as explained later in this chapter.

Agriculture's Evolution, in Brief

A quick summary of the evolution of agriculture helps identify some of the key problems needing to be addressed to achieve a sustainable future. In general, Fig. 12.1 illustrates that over the history of mankind there have been three general types of agriculture, with a sustainable agriculture being the next type to evolve. In the beginning, man was a hunter-gatherer. Agriculture followed as a more-stable system for meeting the food needs of people and, as it became established, agriculture enabled people to develop villages and become less migratory. Traditional agriculture, the first type to develop, was 'sustainable' in that people could use only what natural resources were available within a short distance and that system did not harm the environment over time. In essence, it was a system of raising plants and animals in a convenient place. The only real improvements in this first type of production system compared with what was naturally occurring in the area was the human input that assured water availability and, gradually, the elimination of competing plants from the area being cultivated (although 'slash and burn' systems prove not to be sustainable). Thus, productivity levels for traditional agriculture were only slightly higher than the yields offered by nature. Eventually, the current total output capacity of an area would become insufficient to meet the demands of the expanding population, thus forcing a change in the production system in use.

For most of mankind's history the only real change needed to meet higher levels of demand was to expand the total area being cultivated. However, in the late 19th and early 20th centuries, population growth in many regions made it difficult to keep up with demand for food, even with trade between people within the region. As a result, people began to seek higher productivity levels from the available farmland by adopting 'industrial' production methods. This involved applying many man-made inputs. This began the second type of agriculture, which was viewed as an improvement upon traditional production systems. (Hence, the vertical arrow in Fig. 12.1 indicates an upward movement in the development of agriculture, as producers shifted from 'traditional' to 'industrial' methods – as shown on the bottom of Fig. 12.1.) Industrial agriculture has included mechanical, chemical, biological and managerial revolutions that continue, in varying degrees, to this day. The rate of industrialization in US agriculture increased rapidly during and immediately after the Second World War when, first, large numbers of men were pulled off farms for the war effort and, next, large amounts of capital flowed into farming. The third type of agriculture to evolve, 'organic', has been slowly transitioning into favour as concerns over industrial agriculture's effects on the environment and 'quality of life' have increased in recent decades. Whereas industrial production systems in agriculture have much higher levels of productivity than did earlier systems, they use many inputs that may be harmful and are certainly not sustainable. Organic production systems reduce the use of some harmful inputs, but many of the

Agriculture type	Description	Concerns forcing change
Sustainable	An evolving goal	
Organic	Reduces some harmful inputs	Non-renewable resources, lower yields
Industrial	Higher productivity, inputs not sustainable (renewable)	Ecology, 'quality of life'
Traditional	'Sustainable', low productivity	Total capacity within trade regions

Fig. 12.1. The evolution of agriculture.

systems in use currently have lower yields than the industrial systems they are replacing (Blank and Thompson, 2004) and, thus, cannot sustain the world's population at current levels of demand for food. As a result, 'organic' agriculture is viewed by many production scientists as an interim stage between 'industrial' and 'sustainable' production systems in wealthy nations only. The arrow in Fig. 12.1 points upward to the ultimate goal of achieving a sustainable agriculture.

In this brief sketch of agriculture's evolution, it is clear that the productive capacity of each system is the key concern causing society (including farmers, natural and social scientists, and policy makers) to search for alternative systems. Traditional agriculture is sustainable with regards to its effects on the environment and it is still practised in many less-developed parts of the world. It is being abandoned by those developing countries as soon as they can afford to shift to industrial production systems because additional volumes of food are needed for their growing populations. Thus, one reason developing countries move from traditional to industrial agricultural systems is in response to short-run needs for food. This is the first step up the simple development ladder depicted in Fig. 12.1. Conversely, wealthy countries are moving up, slowly leaving some industrial systems for organic production methods, because of concerns for the long-run effects of industrial production on the environment and the health of their populations. Ultimately, both concerns must be faced by all countries and, it is being argued by increasing numbers of advocates, the final solution to both problems must combine the productivity of industrial systems and the environmental neutrality of organic systems to create a truly sustainable agriculture.

The Economics of US Agriculture

The current state of profitability in US agriculture is very mixed. Many producers across the country are quite profitable, so much so that the summary data published by the US Department of Agriculture appear to indicate that production agriculture is a viable industry. However, a closer look finds that a majority of farmers and ranchers are not truly profitable and that the industry as a whole is far from sustainable (Hoppe and Banker, 2010).

In a detailed analysis of the many questions related to US agriculture's current level of profitability and its struggle for long-term sustainability, Blank (2008) presents three general results that illustrate the challenges facing the industry. First, he notes that economic theory clearly shows that perfectly competitive commodity markets average zero profits over time, thus agricultural producers are always struggling to find more profitable niches as competition squeezes agricultural investment returns below the levels offered by alternatives. Second, the 'technological treadmill' (Cochrane, 1958) keeps US agriculture in the increasingly global markets for commodities by creating new technology that lowers unit costs, but the treadmill is not truly sustainable because it views nature as a competitor to be overcome. And the third general result is that government agricultural policies try to help US producers, but most policies currently focus on subsidizing market profits – especially for Midwestern grains – rather than making those crops more competitive so as to *solve* the underlying problem.

The first of Blank's (2008) results can be summarized by saying that there is a profit squeeze on agricultural producers. Output has outpaced demand for most commodities over the past century, thus leading to falling real prices for the outputs of most producers. On the other hand, the prices of inputs used by agricultural producers have increased dramatically, especially in recent decades. The result is that profit margins are falling, making commodity production less profitable over time and, therefore, a less attractive investment for family farm owner-operators. The decline in profit margins over time is evident in Fig. 12.2. The gross profit margin of US agriculture was about 50% from the beginning of the 20th century through the Second World War (the margin is the share of 'total sales' represented by 'net income', shown in Fig. 12.2), but it gradually declined to about 10% by 2002. Additionally, Blank (2001) shows that average net returns on equity in US agriculture declined from 2.5% in 1960 to 1.5% in 2000.

Blank's (2008) second result is a natural response to the first result: farmers constantly strive to improve their incomes by adopting new

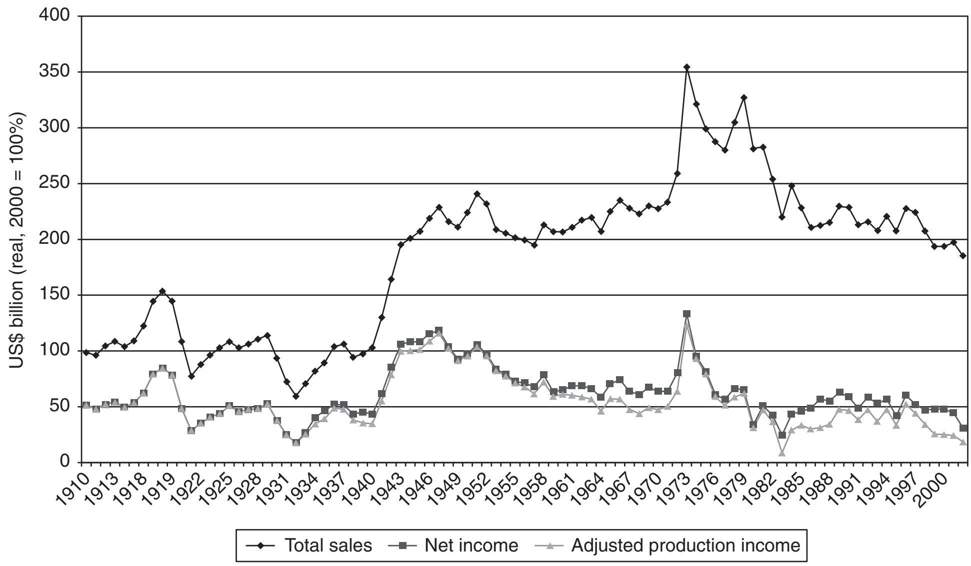

Fig. 12.2. Real US agricultural sales and income, 1910–2002.

technologies. As explained by Levins and Cochrane (1996), ‘early adopters make profits for a short while because of their low unit production costs. As more farmers adopt the technology, however, production goes up, prices go down, and profits are no longer possible even with the lower production costs’. In this era of global markets for most commodities, what is often driving the adoption of new technologies by US producers is the need to compete with foreign suppliers – especially those in less-developed countries – who have lower production input costs. This competition is resulting in increased output, but lower profit margins for producers as they continually search for a profitable niche. That search has mostly involved seeking production methods that give higher yields, despite the long-run effects on the environment – i.e. in the past the environment has been viewed by many as an obstacle to be overcome in the search for greater output.

Blank’s (2008) third result shows which farms receive government payments and where those farms are located. He argues that subsidizing market profits for commodities was a reasonable approach to achieving the goals of aiding the farm economy during the Great Depression of the 1930s and lowering food costs for consumers during the 1940s and 1950s. Unfortunately, those policies led directly to the surplus production that depressed commodity prices and profits over the last few decades thus hurting farmers to the benefit of domestic and foreign consumers of US agricultural commodities. Clearly, a change in focus is needed in US agricultural policy.

In light of these results concerning the profitability of US agriculture, the question arises: ‘how can economic viability be established so the sustainable agriculture goal can be achieved?’ The answer to this important question depends on an understanding of the markets for agricultural commodities, as explained below.

Vertical Coordination in Agriculture

The structure, conduct and performance of US agriculture are continually changing. This may be most easily seen in agribusiness industries where firms are becoming larger and more industrialized, causing industries to become more concentrated. This change in agribusiness’s structure is being driven partly by economies of scale. Conversely, the location-specific nature of agricultural production (which is driven by the comparative advantage of natural

resources and micro-climates) is likely to prevent that industry from becoming as concentrated as agribusiness industries; thus the current imbalance in the bargaining positions of commodity sellers and buyers is expected to get worse in the future. The structural changes leading to concentration, in turn, are likely to change the conduct of commodity markets such that the economic performance of the two industries will be affected, with the agribusiness industry expected to benefit at the expense of the production industry.

One of the ways this change in commodity market conduct is manifesting itself is through the increasing use of production and marketing contracts between agribusiness firms and farmers or ranchers. The trend of increasing contracting was slow to start, but has become more important over the last decade. The overall share of agricultural production value under contract in the US has increased from 12% in 1969 to 39% in 2003 (MacDonald and Korb, 2006). Production and marketing contracts are two methods of vertical coordination. Thus, it has long been hypothesized that the use of these contracts, especially production contracts, is an indicator of industrialization in agriculture (e.g. Mighell and Jones, 1963; Drabenstott, 1995; Ahearn *et al*., 2005).

'Vertical coordination refers to the synchronization of successive stages of production and marketing, with respect to quantity, quality, and timing of product flows' (Martinez, 2002). A production contract offers more control to a contractor than does a marketing contract, but both types of contracts offer only partial control compared with complete vertical integration achieved through common ownership of production and marketing activities at successive stages of the supply chain. A processor firm seeking complete control may prefer vertical integration over the partial control of contracts, *ceteris paribus*. However, farmers and ranchers prefer to be independent operators (Key, 2005) ideally selling their commodities in spot markets, such as auctions. Producers do not like selling in uncertain spot markets, but they prefer competitive spot markets to imperfectly competitive markets in which they are at a disadvantage relative to the buyers they face. Thus, the actual distribution of production being sold in spot markets versus through contracts may indicate (among other factors) the relative market power of market participants.

Contracts formed between agricultural producers and processors replace traditional spot markets for all parties involved. According to results from the USDA's Agricultural Resource and Management Survey (ARMS), contract use is expanding in the USA. The total share of production value under contract has increased from 28.9% in 1991 to 39.1% in 2003. However, there are two different categories of agricultural contracts.

Under marketing contracts, prices, quantities and delivery schedules are agreed upon before crops are harvested or livestock are delivered. Agricultural producers own their commodities throughout the entire stage of production and therefore they retain control over management decisions, including those related to inputs used in production. Katchova and Miranda (2004) found that 'personal and farm characteristics mostly affect the adoption decision rather than the quantity, frequency, and contract type decisions'. Marketing contracts cover a greater share of crop production than livestock production, with 29.7% of total crop production value under marketing contracting compared with 13.7% of livestock production value in 2003. For all commodities produced in the USA, the total share of production value under marketing contracts has been about 21% since 1994 (MacDonald and Korb, 2006).

Under production contracts, the commodity buyer sets specific input specifications and typically provides inputs such as veterinary services, feed and young animals in the case of livestock. In some cases, the buyer owns the commodity being produced from the beginning of the contract period and has managerial control over the production process. In all cases, the producer provides technical and managerial inputs plus all labour and physical facilities needed to create the specified output. Additionally, the producer's payment is not agreed upon prior to the harvest/delivery but rather is determined at the end of the arrangement and is based on quantity and the degree to which the final product meets the buyer's specifications. Production contracts are much more prevalent among livestock commodities than they are among crops. In 2003, only 1.1% of total crop production value was under production

contracts, compared with 33.7% for livestock. Furthermore, the share of total US agricultural sales under production contracts increased from 10.6% in 1996 to about 18% in 2003, in contrast to the stable trend in marketing contracts (MacDonald and Korb, 2006). Table 12.1 summarizes the share of production under contract by commodity and contract type for recent years. Given that producers lose some of their autonomy under the terms of production contracts, their choosing these contracts over spot markets is somewhat surprising, thus justifying a quick review of producers' motivation.

Producer Motives for Production Contracting

Over the last five decades, the literature has offered a fairly consistent list of motives for agricultural producers to choose contracting, but there has been no consistency in opinions of which motives are most important. In 1963, Mighell and Jones identified four reasons for coordinating by non-market means: to increase efficiency, to obtain (or reduce the cost of) financing, to reduce uncertainty, and to gain market advantage. Ahearn *et al.* (2005) said the two most commonly cited reasons for entering

Table 12.1. Share of total agricultural sales by commodity, contract type and year, 1991–2003. Sources: MacDonald and Korb (2006) and the USDA's Agricultural Resource Management Survey for relevant years.

	Share of total sales (%)					
Item	1991–1993	1994–1995	1996–1997	1998–2000	2001–2002	2003
Commodities produced under marketing contract						
All commodities	17.0	21.2	21.5	20.4	19.7	21.7
Crops	22.8	24.0	21.1	22.5	24.7	29.7
Maize	10.2	13.8	12.9	12.6	14.7	13.8
Soybeans	9.6	9.8	13.2	9.7	9.5	13.6
Wheat	5.8	6.2	9.0	6.9	6.4	7.5
Sugar beet	88.5	83.7	74.6	83.1	95.8	95.1
Rice	19.7	25.2	25.8	30.5	38.6	51.8
Groundnuts	45.2	58.3	34.2	44.9	27.9	53.3
Tobacco	0.3	0.6	0.3	1.9	52.6	54.8
Cotton	30.4	44.4	33.8	42.9	52.6	50.9
Fruit	N/A	61.0	54.3	63.3	60.1	67.2
Vegetables	N/A	45.3	32.3	27.3	31.5	36.4
Other crops	6.3	14.0	18.7	21.2	30.9	44.7
Livestock	11.6	18.2	22.0	18.4	14.5	13.7
Broilers	5.9	3.4	4.0	3.9	4.2	1.1
Hogs	N/A	2.4	2.7	9.1	6.1	6.8
Cattle	N/A	4.3	5.9	4.6	2.7	3.4
Other livestock	0.1	6.8	4.9	10.7	3.5	7.4
Dairy	33.6	56.7	58.0	53.4	48.0	50.5
Commodities produced under production contract						
All commodities	11.8	13.0	10.6	16.9	18.0	17.5
Crops	1.9	1.9	1.8	4.2	3.1	1.1
Vegetables	N/A	9.7	6.1	12.4	10.6	6.3
Livestock	21.1	24.7	22.9	29.6	33.8	33.7
Broilers	82.8	81.2	80.1	84.9	88.1	95.5
Hogs	N/A	28.7	47.3	76.3	78.1	84.8
Cattle	N/A	14.7	11.1	19.7	18.3	25.4
Other livestock	0.1	2.6	N/A	N/A	5.5	N/A
Dairy	0.2	0.2	0.1	0.2	0.7	0.6

into contracts were risk management and minimization of production and/or transaction costs. These two reasons for contracting are essentially the same as the first three listed by Mighell and Jones (1963), with efficiency gains and financing being lumped under the production-transaction cost minimization umbrella. Some recent studies (e.g. Martinez, 2002; Allen and Lueck, 2003) have focused on the single explanation of transaction cost economics (Williamson, 1979) and its emphasis on asset specificity as the driving force behind the decision to contract. For example, Lajili *et al.* (1997) found 'the degree of asset specificity significantly influences farmers' choices of contractual arrangements'. However, as pointed out by MacDonald *et al.* (2004a), 'one weakness of transaction-cost analyses is that they typically do not nest market power and efficiency explanations'. In Joskow's (1989) summary, they 'frequently ignore the possibility that there may be market power motivations or market power consequences for these organizational arrangements as well'. Surprisingly, Mighell and Jones' (1963) fourth motive cited, to gain market advantage, has received the least research attention although it is argued here that it is the most likely explanation in US agriculture's current evolutionary state.

Gaining a market advantage may be easy in an industry like agriculture, which has imbalances in its structure (such as having many sellers and few buyers of a commodity). For example, Lanzillotti (1960) detailed how firms dealing with agriculture were already taking advantage of the production sector. He concluded that 'leading firms possess considerable market power and are inclined to utilize such power to manage or administer their market situation'. The result of that market power imbalance was a significant difference in the profit margins of agribusiness firms and agricultural producers. In other words, gaining market power facilitates taking actions that improve a firm's profit margins, thus providing the strongest of incentives to seek bargaining power. As a result, it is surprising that relatively little empirical research was done to sort out the relationship between industry structure and market power. By 1986, the story was still unsettled, as reported by Schrader (1986):

> The relation of integration or nonmarket vertical coordination to market power has two interpretations. Integration and contract coordination are viewed by some as a means to enhance the integrator's market power. Others see market power on one side (or both sides) of a market as an incentive for vertical arrangements to capture gains from the side possessing market power or to achieve joint profit maximization.

The uncertainty was still apparent in 2005 when Ahearn *et al.* (2005) reported on the increasing concentration in agriculture and agribusiness, and noted that 'it is not obvious whether this concentration is the desirable result of cost efficiencies in production or the undesirable result of market power on the part of various players in the supply chain', citing the question raised by Williamson (1968). Thus, more research is needed on the influence of agricultural market structure on conduct such as contracting.

There is little literature dealing directly with the recent rise in production contracting. This is due partly to the scarcity of data on contracting (Ahearn *et al.*, 2005). A review of the scant literature points to three possible explanations for the increased share of production under production contracts. These are risk aversion, the increase in processor concentration in US agribusiness, and the increase in the total scale of agricultural production. While risk management is virtually undisputed in the literature as a catalyst for contracting in general, MacDonald *et al.* (2004b) and Key (2004) stress that it should no longer be considered the sole motivating factor for farmers and ranchers in choosing production contracts. The respective causal relationships between the increase in processor concentration and the increase in the scale of production with production contracting are less clear, but it is proposed here that concentration and size lead to market power that is used to expand contracting.

A defining characteristic of the ongoing transformation of US agriculture may be the rise in concentration in the food manufacturing industry (Ollinger *et al.*, 2005). According to data from the USDA, the mean industry four-firm concentration ratio (CR4) in food manufacturing has risen from 35% in 1982 to 46.1% in 1997 (US Department of Commerce, 2006; CR4 is the concentration ratio measured using sales data from the four largest firms in the industry.

It is the percentage of total industry sales revenues that are accounted for by the four largest firms. CR8 and CR20 are also used in some analyses). The rate of increase in concentration for the meat-packing industry, in which there is also the highest degree of production contracting, significantly outpaced agriculture as a whole. The meat-packing CR4 increased from 29% to 57% over this period. This trend continues in various processing industries. For example, the CR4 of US beef packers was estimated at 81% in 2002 and the CR4 for pork packers in 2002 was found to be 64%. Table 12.2 presents CR4 data for a cross-section of commodities over time.

Given that commodity producers have a strong preference for autonomy (Key, 2005), the observed increase in processor concentration suggests that bargaining power on the part of agricultural producers is decreasing, thus fuelling the trend in production contracting. This certainly appears to be the case in the hog industry where producers who value autonomy less than they fear the risks of being without a contract eagerly adopt contracts (Davis and Gillespie, 2007). However, there are exceptions to this argument. For example, the soybean processing industry saw an increase in concentration from 1982 to 2002, yet only a small portion of total soybean production is under any form of contract, as indicated in Table 12.1. The broiler industry has by far the largest share of production under production contract, yet among livestock commodities it has both the lowest CR4 and the slowest growth in concentration over the comparable time.

Producer concentration is also on the rise in US agriculture. According to USDA data, the percentage of farms in the USA with annual sales of US$500,000 or more has increased from 2% in 1991 to 4.4% in 2001. More strikingly, these farms' share of total agricultural production increased over this period from 39% to 57.4%. Examining individual commodities, Rios and Gray (2005) determined that the share of industry total sales from farms with annual sales of US$500,000 or higher increased from 10.9% to 77% for hogs from 1982 to 2002. Production contracting is relatively very high for hog production, even though the rate of growth in hog producer concentration significantly outpaced the equivalent numbers for commodities with low production contracting, such as wheat, maize and soybeans. Just as increased processor concentration implies increased buyer bargaining power, increased producer concentration would normally suggest increased seller bargaining power. However, concentration of hog producers may be an outcome caused by the trend of processors offering contracts most often to larger producers only. Thus, the hog industry case indicates there are some commodity-specific factors influencing the level of production

Table 12.2. Commodity industry concentration (%).

Commodity	CR4 1987	CR4 1992	CR4 1997	CR4 2002
Broilers	29	34	56	54
Hogs	20	25	64	68
Cattle	39	50	84	86
Dairy	21	22	21	30
Soybeans	71	71	75	95
Maize	74	73	80	69
Wheat	44	56	62	49
Oats	27	33	64	70
Barley	19	23	46	87
Rice	41	51	69	57
Cotton	18	19	20	26
Sugarbeet	83	85	85	85
Groundnut	68	80	82	87
Tobacco	70	76	83	89

Notes: 'CR4' is the concentration ratio reported by the US Census Bureau for the major product category for the year indicated. The source for the CR4 and for the data used in the usage index calculations is the Census Bureau's (US Department of Commerce, 2006) *2002 Economic Census*.

contracting and the direction of causality in that contracting (Key and McBride, 2003).

Due largely to the location-specific nature of agricultural production, the food manufacturing sector is likely to consolidate faster than the commodity production sector. That is what happened in the UK (Duranton and Overman, 2005). However, concentration in the US manufacturing industry is not the primary determinant of the pattern of production contracting, particularly when considering the current trends in producer concentration. Clearly, many factors are significant, as noted below.

Key (2004) examined the supply side of agribusiness by evaluating the relationship between the scale of production and contracting. The scale of production, as measured by changes in the size and output of the largest farms by sector, was found to be directly correlated with the prevalence of contracting. Explanations offered by Key (2004) for this correlation included the usual stories of grower risk aversion and contractor transaction costs, as well as newer theoretical justifications such as asset specificity.

Finally, another possible determinant of contracting is the growth of production contracting itself. Recent research suggests that farmers in some commodity markets are turning to contracting out of necessity due to the incomplete markets created by other market participants' decision to contract (Young and Burke, 2001). Roberts and Key (2005) demonstrated that in some markets, farmers who choose to engage in production contracts could impose negative externalities on other farmers in the form of increased search and transaction costs. The farmers facing the externalities are induced to enter into contracts, which they would not have done otherwise, because contracts may represent the only available access to a buyer. This finding is consistent with the idea that spot markets have 'tipping points' at which a market is thinned enough to induce all remaining participants to enter into contracts (MacDonald *et al.*, 2004b).

It is clear from the literature that questions still remain as to the primary determinants of production contracting in agriculture. Also, much is yet unknown regarding the effects of contracting on producers, agribusiness and consumers. Yet, it is understood that contracting has played a large role in improving product consistency and traceability throughout the stages of food production (MacDonald *et al.*, 2004b). Furthermore, research has shown that contracting has a positive effect on farm productivity (Ahearn *et al.*, 2002; Key and McBride, 2003; Morrison *et al.*, 2004). There remain concerns over the effects on producers who enter into contracts against their best interests (Roberts and Key, 2005), and the managerial control imposed on producers by the processors with whom they contract (Farm Foundation, 2004). However, much of the rise in production contracting has occurred in just the past decade, suggesting that it may take years for the large-scale effects of production contracting to become evident in empirical analyses across a wide range of commodities.

Farm-level Analysis

Several hypotheses about the influence of production contracting on the size, structure and financial position of production operations are tested here using farm-level survey data. Producers who have production contracts are compared with those who remain independent. Based on the literature (e.g. Key, 2004; Morrison *et al.*, 2004; Roberts and Key, 2005), it is hypothesized that producers entering into production contracts are likely to be larger than independents, significantly less diversified in terms of commodities produced, and facing increased risk, relative to the risk exposure of independents. Independent-sample *t*-tests of these and related hypotheses are conducted for a cross-section of commodities. Using pooled farm-level data from the USDA's Agricultural Resource Management Survey for the years 1996–2004 (USDA/ERS, 2004) gives a total of 95,517 observations.

The share of total sales under production contract varies greatly among commodities in the USA. Fourteen major US commodities for which adequate data were available are examined and it is found that a continuum exists with regards to production contracting, ranging from virtually all production being under contract for broilers to no production contracting in the case of tobacco. Also, previous research has found significant differences between firms that enter

into production contracts and those that remain independent (Key, 2004).

A small sample of commodities was evaluated in more detail to enable formal tests of hypotheses about differences in farm characteristics between production contractors and independent producers. Table 12.3 presents various statistics, by commodity and the results of independent-sample *t*-tests of differences in the reported average values for the two groups. Several patterns appear across the results, as described below.

The first hypothesis tested is that production contractors have a higher per farm output of the relevant commodity than do independent operators. The results are shown in the two rows labelled 'sales of the commodity' in Table 12.3. The values are the annual average sales of only the commodity of interest, not total farm sales. For example, of the operators surveyed who produce broilers, those with contracts covering broiler production averaged US$675,979 in broiler sales annually from 1996–2004. In contrast, independent broiler producers sold only US$27,513 worth of the commodity annually, on average. For all of the commodities listed in Table 12.3, contractors produce significantly greater quantities per farm than do independents, on average. Also, in each case the *t*-test indicates that the difference in average sales is statistically significant, thus supporting the hypothesis. One implication of this result is that having a production contract may encourage operators to expand the scale of their output of the contracted commodity, although the direction of causality could be the reverse; producers who want to go large-scale adopt contracts to share risk, reduce transaction costs and share managerial responsibilities. The risk-reducing character of production contracts may enable producers comfortably to expand their operations to achieve economies of scale. For example, Key and McBride (2003) found that for hog producers the use of production contracts is associated with a substantial increase in factor productivity, and represents a technological improvement over independent production.

The result above leads to a second hypothesis, that firms with production contracts will be more specialized, less diversified, in their commodity output. Diversification is a tool used by producers to reduce risks, so the implication is that having a production contract substitutes for diversification as a risk management tool. In Table 12.3, the commodity share of total sales is used as a measure of specialization. For all the commodities listed, contractors get a higher share of their total sales from the contracted commodity. As anticipated, livestock contractors are significantly less diversified than are independent producers. Moreover, as the percentage of producers engaged in production contracts increases among livestock commodities, the degree of diversification decreases. These results support the hypothesis, especially for livestock producers.

The limited data available here do not make it possible to directly test whether livestock producers are yielding net economic benefits from production contracts. However, the statistics in Table 12.3 show that among livestock commodities average total income and average farm net worth for contractors decrease in both absolute terms and relative to independent producers as the share of production contracting increases and diversification decreases. Both broiler and cattle producers earn the majority of their total household income off the farm, in contrast to independents. The debt-to-asset ratio is a commonly used measure of financial risk for producers and livestock contractors have a significantly higher ratio than do independent producers. In general, these results indicate that livestock operations using production contracts are larger, but less profitable, than independent operators and face slightly more financial risk. However, these observations vary inversely with the physical size of the animal involved, applying most strongly to broilers and to a lesser extent to hogs and then cattle.

Crop producers using production contracts are less diversified than are independents, on average, but the differences between the two categories of producers are smaller in the case of crops than they are between livestock producer categories. Also in contrast to the relationships governing livestock production, crop contractors typically have significantly greater household income and net worth than do independents, plus significantly smaller shares of income coming from off-farm sources.

The most readily apparent difference between the livestock and crop commodity markets is that production contracting is a less

Table 12.3. Production contracting in US agriculture, Summary of average results per farm, 1996–2004. Data source: the US Department of Agriculture's Agricultural Resource Management Survey for the years 1996–2004.

	Commodity					
	Broilers	Hogs	Cattle	Maize	Soybeans	Cotton
Total number of producers surveyed	4,713	6,620	50,166	27,852	29,770	6,427
Farmers who Production Contract (%)	86.3	25.8	1.04	0.36	0.52	0.09
Contracting share of commodity sales (%)	95.5	78.7	18.6	0.77	1.44	0.12
Sales of the commodity, Contractors (US$)	675,979***	753,164***	631,546***	201,558***	130,994***	373,125***
Sales of the commodity, Independents (US$)	27,513	70,979	29,023	60,171	46,772	159,864
Total farm sales, Contractors (US$)	909,943***	1,329,973***	2,839,963**	558,902**	528,445**	720,208
Total farm sales, Independence (US$)	626,224	435,290	395,561	458,739	453,176	682,714
Commodity share of total sales, Contractors (%)	74.3***	56.6***	30.5***	36.3***	24.8	51.8***
Commodity share of total sales, Independents (%)	4.1	15.6	7.3	8.8	17.2	23.4
Total household income, Contractors (US$)	71,003***	104,172	158,879***	166,548**	125,191*	410,229*
Total household income, Independents (US$)	190,669	99,924	86,189	96,204	101,333	158,648
Off-farm share of income, Contractors (%)	58.2***	33.4	53.04***	28.3**	26.3*	6.4*
Off-farm share of income, Independents (%)	23.3	39.4	21.4	40.3	38.5	27.2
Farm net worth, Contractors (US$)	698,145***	894,956	981,894	1,220,000*	1,010,000*	2,180,000*
Farm net worth, Independents (US$)	899,987	940,565	975,049	939,469	882,686	922,669
Debt-to-Asset ratio, Contractors	0.26***	0.24***	0.31***	0.22	0.26**	0.08
Debt-to-Asset ratio, Independents	0.14	0.18	0.18	0.31	0.53	0.17

***, **, * indicate a statistically significant difference between the mean values for producers who contract versus independent producers at the 99%, 95% and 90% confidence levels, respectively.

popular choice among crop producers, as noted in the existing literature. Among all the crop commodities in Table 12.3, the percentage of farmers using production contracts is less than 1%.

Crop contractors produce significantly greater quantities of the commodities contracted than independents, as was true in livestock markets, but the average differences are considerably smaller in magnitude. Among crop contractors, commodity sales exceed those of independents by 55.5% on average, while the equivalent margin for livestock producers is 94%. In turn, crop contractors are more specialized than are independent crop producers, but crop contractors are more likely than livestock contractors to rely on some combination of contracting and diversification to manage risk.

Formal hypothesis testing on the financial net benefits of contracting is not possible with the limited data available, but our preliminary empirical results suggest that crop contractors reap greater benefits from production contracting than do livestock contractors. This may reflect the difference in producer bargaining power in livestock versus crop markets, with crop producers having more products made from their commodity, thus having more buyers available to them than do livestock producers. Risk, as measured by the debt-to-asset ratio, appears to be a significant motivating factor in favour of using production contracts in the case of livestock producers, but the same cannot be said for crop producers. Finally, these and other circumstances have changed across commodity markets over the past decade as markets have become increasingly concentrated, especially within the livestock sector. Thus, this study has raised many hypotheses to be tested in the future as more data on production contracting become available.

Hog Case Study Results

The hog industry has been the subject of much research on changes in livestock industry structure and the trend toward increased production contracting (e.g. Martin, 1997; Bessler and Akleman, 1998; Martinez, 1999; Key and McBride, 2003; Ollinger *et al*., 2005). Thus, it is used here to illustrate the relationship between livestock production contractors and independent producers.

The most important observation is that over the period 1996–2004, the percentage of hog producers using production contracts has increased steadily. The share of total hog production under contract increased even more drastically, reaching 87% in 2004. As production contracting increased in scale, the diversification of the hog contractors decreased steadily, both in absolute terms and relative to independent hog producers. This means that the shift in hog industry structure toward most farm-level output being under production contract appears to have had the effect of substituting contracts for diversification as a risk management strategy for most hog producers. This may partly explain why production contractors had higher debt ratios than independents over most years. Although the data cannot answer the question of whether contractors have higher debts because they think that contracts reduce their financial risk exposure, or whether the higher debt ratios reflect the higher capital requirements of a larger, more-specialized hog operation, it is expected that both explanations are partly accurate.

The financial performance data available contradict the hypothesis that contracts reduce producers' financial risk exposure. During most years in the 1996–2004 period, contractors and independent hog producers were statistically equivalent in terms of average Net Farm Income. However, despite significantly higher sales of hogs and total farm sales, the average farm net worth of contractors has never significantly exceeded that of independents. Thus, production contracts have not led to higher wealth. Also, the fact that hog production contractors are steadily decreasing their share of off-farm income, indicates that the larger scale of operations needed under contract has led to more specialized hog operations, leaving less time for off-farm income opportunities. This combined degree of household income specialization may give contract hog producers a *higher* degree of financial risk exposure than that faced by independent hog producers. This is apparent when comparing the standard deviations of the average Net Farm Income over the 9 years: it is US$48,260 for contractors and US$15,016 for

independent hog producers. Thus, the structural change that has led to increased production contracting has not significantly improved contractors' income, compared with independent operators, but it may have increased their exposure to income risk. Therefore, hog production contractors may be worse off financially, on average. This raises the interesting question: do hog producers accept contracts because they think the productivity improvements found by Key and McBride (2003) will lead to improved profitability, or do they generally consent to the contract because they do not have the bargaining power to resist the demands of their buyers, as implied by Davis and Gillespie (2007)?

Implications of the Contracting Results

The preliminary empirical results here generally show that production contracts lead to production specialization, which, in turn, may reduce off-farm income opportunities, both of which can increase the income risk of producers. This is an important observation because it contradicts one of the main arguments used to justify production contracting. Proponents of contracting and much of the theoretical literature have said that producers can use contracts to reduce risk, which is true. For the small cross-section of commodities evaluated here, the reality is that contractors have higher sales totals and higher income variance than do independent producers, but not necessarily higher income levels, on average.

It has been argued in the literature that buyer bargaining power increases with industrialization and that the potential for industrialization is influenced by a commodity's physical attributes (e.g. Sheldon, 1996). In particular, it has been well established that livestock processing industries have scale economies that encourage continued industrialization and that the resulting industry concentration of the last few decades has facilitated increased use of production contracts in those markets (Drabenstott, 1995; Morrison Paul, 1999, 2000, 2001; Key, 2004; Ahearn *et al.*, 2005; Bhuyan, 2005; MacDonald and Korb, 2006). In crop industries, however, production contracting is rare in most markets, although marketing contracts cover a majority of output in some markets (MacDonald and Korb, 2006). These differences across commodity types were apparent in the analysis here and raise questions for future research.

Looking to the future, the results of this preliminary study indicate that production contracting is likely to continue expanding to cover a higher share of total output for many commodities. This is an incentive for producers to form cooperatives or to use some other type of collective selling arrangements. However, cooperatives, bargaining associations and other selling arrangements employ a type of production contract with supplier-members. Therefore, all trends indicate it may be increasingly difficult for producers to maintain their independence in the industrialized agriculture of America's future.

Blending Animal Agriculture and Agribusiness for Success

Thus far, this chapter has presented a picture that is bright for the US agribusiness sector, but bleak for the agricultural production sector. However, this is not the end of the story. Both sectors can survive in the future if industry participants take a slightly different perspective when viewing those in the other sector. It is argued in this section that blending US agriculture and agribusiness may be essential for success in the future (especially for the production sector) but, if accomplished, the resulting agri-food industry will play the leading role in the global market. In doing so, the new industry can create a truly 'economically sustainable agriculture' in America, whereas none exists currently without policy interventions.

To begin, the concept of 'blending' agricultural production and agribusiness is described. A blended industry, in the simplest sense, is one in which all participants understand and appreciate their mutual dependence on all other participants. No matter what form of vertical governance is used to blend firms into a coordinated system, the key point is that everyone in the system knows that it will fail without the contributions of each participant. Thus, everyone knows that their economic rewards depend in part on the performance of others in the system.

Existing examples of a blended industry include the horticulture-nursery and the dairy-milk industries. In the first case, the horticultural participants are farmers producing plants that the nursery participants sell through wholesale and retail outlets. Without the plants, the nurseries have nothing to sell, and without the nurseries, the farmers have no market outlet. Each group needs the other. In the second case, dairy farmers produce raw milk that is processed, packaged and distributed by the second group of participants. Again, each group needs the other. As a result, there is much communication and cooperation between the groups. The first group seeks to deliver a product that facilitates the input needs of the second group. That is possible because the second group carefully communicates its needs to the first group. In essence, the groups *try* to blend their activities into a seamless whole that has the best chance of successfully meeting the demands of consumers.

These two examples of a blended industry are similar in that the product's form is changed little in a vertical system that is 'short' from top to bottom. In this short vertical system, it is relatively easy for participants to both see how the other group contributes to the whole and to communicate with each other. However, in the future, some blended industries must be very 'tall' to serve their product markets, thus making it much more difficult for system participants to recognize and appreciate the contributions of all other participants. This is the challenge driving current market evolution.

The Current Situation

What is the current situation in the market evolutionary process shaping US animal agriculture and agribusiness? In simple terms, America is at a turning point between two eras in the relationship between its production agriculture sector and its agribusiness sector. The first era is not yet over, but will be soon. What will end the first era, and what will differentiate the second era from the first, is a simple change in the perspective of industry participants toward members of the other sector. At present, both groups need the other, but they are in a 'tug-of-war' when interacting, each seeking to maximize their own profits. This is a state of conflict, which is not sustainable.

Structural changes in US agricultural production are occurring in response to the increased globalization of commodity markets. Boehlje (1999) summarizes the changes by saying 'production is changing from an industry dominated by family-based, small-scale, relatively independent firms to one of larger firms that are more tightly aligned across the production and distribution value chain'. These changes are occurring against the wishes of many producers. As Key (2005) indicates, agricultural producers are very independent people, thus not eager to give up any control over their operations, if possible. Yet, that is what is happening at present. The agribusiness sector is using its market power to nudge producers into production and marketing contracts. As a result, there is lots of conflict in the interactions between the production and agribusiness sectors. 'Some would argue that the basic nature of competition has changed in all industries in recent years, especially in terms of the definition of a market' (Boehlje, 1999). He adds, 'worldwide sourcing and selling has changed the geographic boundaries of markets from regional or national to global'. In response, 'closer vertical coordination has occurred as the use of spot markets has declined, while production and marketing contracts, franchising, strategic alliances, joint ventures, and full vertical integration have increased' (Young and Hobbs, 2002).

This evolutionary change in markets for commodities makes it more difficult for independent farmers and ranchers to access buyers in a traditional negotiation, thus adding to the pressure on producers to align themselves with some new vertical coordination structure. Up to this point in time, most US producers have viewed these market changes as a threat. That perspective is understandable given the negative effects market changes have had thus far on producers' financial performance. However, that perspective could be the downfall of US agriculture.

The irony of the current situation is that if nothing changes current perspectives, the threat to producers posed by being forced to join a supply chain may be exceeded only by the threat of *not* being able to join a supply chain. As Young and Hobbs (2002) conclude:

> some producers may have difficulty gaining entry to tightly coordinated supply chains. Entry may be difficult due to requirements for

> sophisticated production skills or the need for specialized equipment or capital. The inability of certain producers to gain entry to supply chains for these reasons would be a continuation of the forces that have prompted producers to exit from agriculture historically.

Why would a producer have to exit from agriculture if he or she cannot join a supply chain? The answer is that as more supply chains develop in the future, the fewer participants there will be in traditional spot markets, thus those markets will erode and eventually disappear. In other words, spot markets are becoming thinner, which means they may be less likely to generate the competitive market prices needed to attract participants.

How the Production Sector Can Survive

In the long run, the survival of most US agricultural producers may depend on their willingness to be a contract supplier to an agribusiness that is successfully meeting consumers' demands for specific product attributes. More specifically, for the US production agriculture sector to survive in a future that will be full of new foreign competitors that have lower production costs, US producers will have to voluntarily blend with agribusiness in a 'metasystem' aimed at improving the profits of each participant by improving the competitiveness of the US agri-food firm versus foreign competitors. This strategy does not guarantee the survival of any particular firm or industry, but it is the only approach that adequately addresses the challenges faced by US agricultural producers and, thus, it offers a chance for prosperity.

The first challenge is the current state of conflict between producers and the agribusiness firms with which they deal. As long as agricultural producers view agribusiness as part of the problem, rather than as part of the solution, the conflict will continue and more producers will be forced to exit agriculture. On the other hand, if producers follow the old cliché, 'if you can't beat 'em, join 'em', and replace the conflict with collaboration, they immediately raise their chances of survival. This is possible because market structures based on truly voluntary participation will be more successful in the long run because they eliminate internal conflict.

A metasystem is a state of collaboration that helps address the second challenge faced by US agricultural producers: foreign competition. By design, metasystems add value to commodities and differentiate them from the output of competitors. A metasystem is a special type of supply chain. As noted earlier, a 'supply chain' is an integrated vertical system across different functions in the process required to create and deliver a product to the consumer. Most metasystems focus on quality management. Caswell *et al.* (1998) say that food quality metasystems are strategies that affect any quality attribute involving food safety, nutrition, value, packaging or process. The authors reported that 'metasystems are implemented through metastandards, which most often define a process to be undertaken by a company to assure quality on an on-going basis'. Thus, a metasystem is an organized attempt to create and document quality differences in products. All firms in a metasystem willingly collaborate in this effort.

There are many benefits to participation in a metasystem. For example, Caswell *et al.* (1998) state:

> in addition to affecting operation of the value chain, food quality metasystems are likely to confer significant marketing advantages on companies in selling to final consumers. These advantages come from selling a higher quality product and reliably being able to certify that quality to consumers who are willing to pay more. These advantages may enter the company's profit performance through a higher price or lower transaction costs.

This ability to differentiate products based on higher quality attributes is a key weapon in the current conflict between US firms and the growing number of foreign competitors. As US producers lose the race to be the lowest-cost supplier of commodities, their salvation rests in being identified as the supplier of high-quality commodities as inputs to US agribusiness firms that create high-quality consumer products.

Metasystems are the future for the US food industry. For example, Fouayzi *et al.* (2006) found that over 90% of fresh-cut produce firms have adopted a quality management system because, among other reasons, it facilitates trade between firms. With such a system, long-term contracts are more likely between firms within a supply chain, and transaction costs are reduced.

The ability to make long-term contracts holds great value for producers in many commodity markets. For example, it would help reduce the chances of being held-up by processors – a major source of conflict in the current relationship between many producers and agribusiness. As Vukina and Leegomonchai (2006) explain, when only short-term contracts are available, commodity producers can be held-up by processors because:

> growers' assets are a source of potentially appropriable quasi-rents in the sense that they have low salvage value outside the bilateral contractual relationship. This constitutes a hold-up problem that can manifest itself in two ways. First, … appropriable quasi-rents affect the level of investments. Being aware of the possibility that they may be held-up by processors, growers will cautiously invest in specific assets. [Second, after] facilities have been constructed, the processor may exploit his advantageous bargaining position by frequently requesting upgrades and technological improvements as conditions for contract renewal.

As a result of the hold-up risk, producers underinvest in assets with specific uses (Castaneda, 2006). A long-term contract reduces the risk of hold-up and, in the process, reduces the state of conflict between producers and food manufacturing firms. The increased state of collaboration encourages producers to invest in more assets with specific uses, thus providing expanded output to agribusiness without those firms having to increase the number of contracts negotiated or supplier relationships maintained. This reduces transaction costs to all parties involved. The ability to sign long-term contracts also gives supply chain participants the ability to adopt many practices aimed at gaining a competitive advantage over other firms, such as time integration (Wilson and Thompson, 2003) and other innovations. However, at present it is usually agribusiness firms resisting the move to long-term contracts (e.g. in the broiler industry), so they are apparently not yet willing to accept the advantages of long-term contracts and move to a full metasystem.

In total, the economics of supply chains and their effects on the structure of agriculture seem to be positive for agribusiness and consumers, so at this point in time there is no reason to think their growth will slow. Given this clear trend for agribusiness firms, agricultural producers need to decide sooner, rather than later, to join the team and enjoy the perks. Remaining in a state of conflict with agribusiness is a losing proposition. Unfortunately, the conflict may benefit agribusiness firms in some industries, so the path to integration will be bumpy.

The resistance of independent producers to joining a metasystem, or any other vertical market structure, is expected to continue for some time. To avoid dependence on an agribusiness, some farmers and ranchers will continue to pursue the creation of their own supply chains, in the form of direct marketing to a niche market. In some places where large numbers of consumers are located close to talented producers, niche markets will survive and generate adequate returns. In other places, potential niches are simply located too far from the producer entrepreneur to enable the establishment of profitable operations. And finally, niches will fail in lots of places because the producer did not realize that creating a supply chain meant that he would have to perform all the business and production functions himself. Sometimes when a producer talks about 'eliminating the middleman', it is because that producer does not appreciate that agribusiness firms exist because they add value to commodities and it is the processed product that consumers want, not just the commodity which was used as an input in creating the final product. Supply chains create a synergy; the sum is (in) greater (demand) than the (demand for the) parts.

Finally, survival of the US agricultural production sector depends on the ability of producers to adjust to a new business structure. Metasystems and other supply chain structures are changing the theory of the firm. As early as 1992, Barry *et al.* (1992) observed that 'the needs for farm-level product differentiation put pressure on open market relationships and may lead to vertical integration or contracting between key stages in the market system'. Farm-level product differentiation often requires specialized equipment, creating asset specificity and asset specificity, and vertical coordination are considered to be positively related. 'Greater asset specificity means greater transaction costs in redeployment, and a tendency toward more complex, long-term contracting and vertical integration' (Barry *et al.*, 1992). Therefore,

contracting continues to expand, as described earlier, changing the nature of relationships between market participants. For example, production contracts (and other vertical integration tools) create an agency relationship. The agent (producer) is expected to behave in concert with the objectives of the principals (buyers) so that these objectives can be optimally attained. This creates a situation in which 'the manager's task now involves selecting the boundaries of the firm (defined by contractual and asset control relationships) along with the more traditional tasks' (Barry *et al.*, 1992). In other words, US agricultural producers must decide the extent to which they are going to blend their firm with others in a supply chain voluntarily.

A Modest Proposal

Economists can help advance progress toward the goal of a sustainable agriculture by becoming more active in the pursuit of the piece of the goal's definition, which has received almost no attention thus far. The fifth bulleted point in Congress's definition of sustainable agriculture asserts that agriculture affects 'the quality of life for farmers and society as a whole' yet 'quality of life' attributes have largely been ignored by economists. Although the help of other social scientists is needed to identify and define what those attributes might be (both negative externalities and positive amenities), economists can contribute by applying their understanding of markets. Specifically, the most likely answer to the question of how economic viability can be established for a sustainable agriculture is that government policies can assist markets in establishing the value of 'quality of life' attributes, and the assistance of economists will be needed for that to happen.

Quality of life attributes, such as agriculture's effects on health issues and environmental quality, do not yet have values determined in markets. This 'market failure' justifies government intervention. Such interventions are beginning to happen in Europe and it is being discussed by some US agricultural groups as a new approach to agricultural policy. Examples of indirect attempts to recognize agriculture's contributions to, and effects on, the quality of life in the USA include the Conservation Reserve Program, laws limiting agricultural pollution, and local ordinances recognizing and protecting the amenities provided by a rural landscape. A number of environmental protection and/or conservation programmes have been introduced in farm bills since Congress defined 'sustainable agriculture' in 1990. The examples listed here demonstrate the wide range of issues addressed and approaches used by government.

'Multifunctionality' is the label being used for the new theme in agricultural policy debates. The term refers to the existence of multiple commodity and non-commodity outputs that are jointly produced by agriculture (Randall, 2002). Cahill (2001) noted that 'food security, food safety, animal welfare, cultural and historic heritage values, environmental quality, landscape, biodiversity and rural development are just some of the outputs claimed to belong to the multifunctionality of the agricultural sector'. Proponents argue that government intervention is needed because markets do not exist for all of these outputs, especially the non-commodity. If economists and producers can work with policy makers to establish markets for all of the outputs of agriculture, and those markets can establish values for each output, the long-run economic viability of agriculture is sure to change for the better (Paarlberg *et al.*, 2002; Randall, 2002; Smith, 2006). For example, Bennett *et al.* (2004) found Australian urban dwellers are willing to pay some positive amount to maintain rural populations because of the environmental stewardship function performed by rural residents.

Blank (2008) suggests 'America needs to shift the focus of policy from viewing "agriculture as factory floor" to "agriculture as neighbourhood." This change is needed because the farm factory is not always profitable, and taking the "neighbourhood" view helps us realize that agriculture affects everyone.' With this new view, and a multifunctionality perspective, farmers and ranchers can be seen as 'stewards of the land' that are performing a public good that should be valued because it is essential to achieving a sustainable future. Many farmers already view themselves as a steward in a limited sense. Also, Chouinard *et al.* (2008) found that some agricultural producers are willing to sacrifice some profit to engage in stewardly farm practices. A broader notion of stewardship is based in Blank's (2008) call for a new policy perspective.

A multifunctional view would encourage and empower farmers and ranchers as they are formally recognized as stewards of the land/environment, communities, human health and rural vitality.

Such a policy shift would give market values to each alternative identified and/or created by natural scientists (which is not happening at present), such as plant varieties; production methods; systems for improving air, land, and/or water quality; to mention a few. This would point 'what will be' toward an agriculture and an environment that are viewed as seamless components of a single world, rather than as competitors for attention and resources. Animal producers and economists should probably argue to policy makers that this is a fair way to balance the needs of present and future generations to move the evolving world toward sustainability.

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13 Achieving Social Sustainability in Animal Agriculture: Challenges and Opportunities to Reconcile Multiple Sustainability Goals

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Introduction

According to the Consumer Reports 'Greener Choices' eco-label database, there are 56 eco-labels for beef. These labels range from labels for organic certification to animal welfare labels (i.e. humanely raised, free range, grass-fed) to health-based claims including 'no additives' and 'hormone-free'. The website also details 48 eco-labels for dairy, 46 for eggs, 50 for lamb, 49 for pork and 54 for poultry (Consumers Union, 2011). While many of these labels may have overlap among different categories, the dozens of labels now available for products highlights that the public has increasing concerns and desires when they purchase animal products. At the same time, many of these labels offer no substantial means to verify their claims by a third party certifier, leaving consumers increasingly confused about legitimate sustainability claims, and farmers and ranchers increasingly pressured to disclose their farming and management practices.

This trend towards sustainable agriculture and green products as well as a growing interest from consumers about the origins of their food offers new challenges and opportunities to the agricultural industry, farmers, ranchers, farm and ranch workers, policy makers and even the public themselves. Consumer demand for more sustainable animal products is also coupled with concerns about food security, labour and social living standards, and economic viability for farmers and ranchers. Agricultural policies are changing to reflect new standards and niche markets, and agricultural processors, farmers and ranchers are adapting to these new regulations. This chapter will review the many aspects of social sustainability in animal agriculture and discuss the recent challenges and opportunities for sustainability in animal agriculture. The chapter will begin by describing the origins of social sustainability followed by a discussion of social sustainability in animal agriculture. Many aspects of social sustainability will be considered from social–environmental aspects including environmental justice to socio-economic considerations such as fair wages. Finally, the chapter will conclude by considering how multiple sustainability goals can be reconciled and achieved and how policies and consumer demands can influence this process.

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Defining Sustainability – What Do We Really Mean?

Today there are many different definitions for sustainability as defined by companies, governments or individuals. However, the concept of sustainability or sustainable development has its origins at the United Nations. In 1987, the United Nations World Commission on Environment and Development published the Brundtland Report, more frequently known as 'Our Common Future'. In this report, the Commission discussed the challenges facing a growing human population living in a world of finite resources and focused on the opportunities and solutions possible through 'sustainable development'. In an often quoted phrase, the UN Commission stated, 'Humanity has the ability to make development sustainable to ensure that it meets the needs of the present without compromising the ability of future generations to meet their own needs' (WCED, 1987). This definition has continually evolved since its inception as it has been applied to different settings and by various actors and stakeholders. Nevertheless, this concept has been a significant force in guiding concepts of sustainability in the decades since.

Advancing sustainability definitions

It was only 2 years after the UN report that the first published definitions of sustainable agriculture also appeared. In 1989 the American Society of Agronomy stated in a publication, 'A sustainable agriculture is one that, over the long term, enhances environmental quality and the resource base on which agriculture depends; provides for basic human food and fibre needs; is economically viable; and enhances the quality of life for farmers and society as a whole' (Agronomy News, 1989). One year later, the US government (US Congress, 1990) formally defined sustainable agriculture in the 1990 Farm Bill as:

> An integrated system of plant and animal production practices having a site-specific application that will, over the long term …
>
> **1)** Satisfy human food and fiber needs
> **2)** Enhance environmental quality and the natural resource base upon which the agricultural economy depends
> **3)** Make the most efficient use of non-renewable resources and on-farm resources and integrate, where appropriate, natural biological cycles and controls
> **4)** Sustain the economic viability of farm operations and
> **5)** Enhance the quality of life for farmers and society as a whole.

This broad-based definition included many aspects of the three traditional components of sustainability now considered common – environmental, social and economic sustainability (Fig. 13.1). This common trio of sustainability 'pillars' was not however coined until more than a decade later at the 2002 United Nations World Summit on Sustainable Development in South Africa. The conference stated a 'collective responsibility to advance and strengthen the interdependent and mutually reinforcing pillars of sustainable development – economic development, social development and environmental protection – at the local, national, regional, and global levels' (United Nations, 2002). Today, sustainable development is frequently considered in the context of these three pillars, despite earlier definitions of sustainability largely being focused on environmental aspects. As a result, this definition laid the groundwork for advancing concepts of social sustainability and setting goals and targets for achieving it.

Social sustainability as a pillar of sustainability

Early efforts for sustainable development frequently emphasized the concept of environmental sustainability, in part stemming from the environmental activism of the 1960s and 1970s, while economic and social sustainability had been of less focus (McKenzie, 2004). Consumer and society demands and desires related to sustainable animal agriculture production also play a major role in advancements towards sustainability. The large number of eco-labels on the market today indicates a strong desire for environmental sustainability from consumers; certifications and labels for social and economic sustainability measures have been less rapid to enter the market place. One notable example is the creation of fair trade labels that aim to ensure that farmers and farm-workers were paid a living

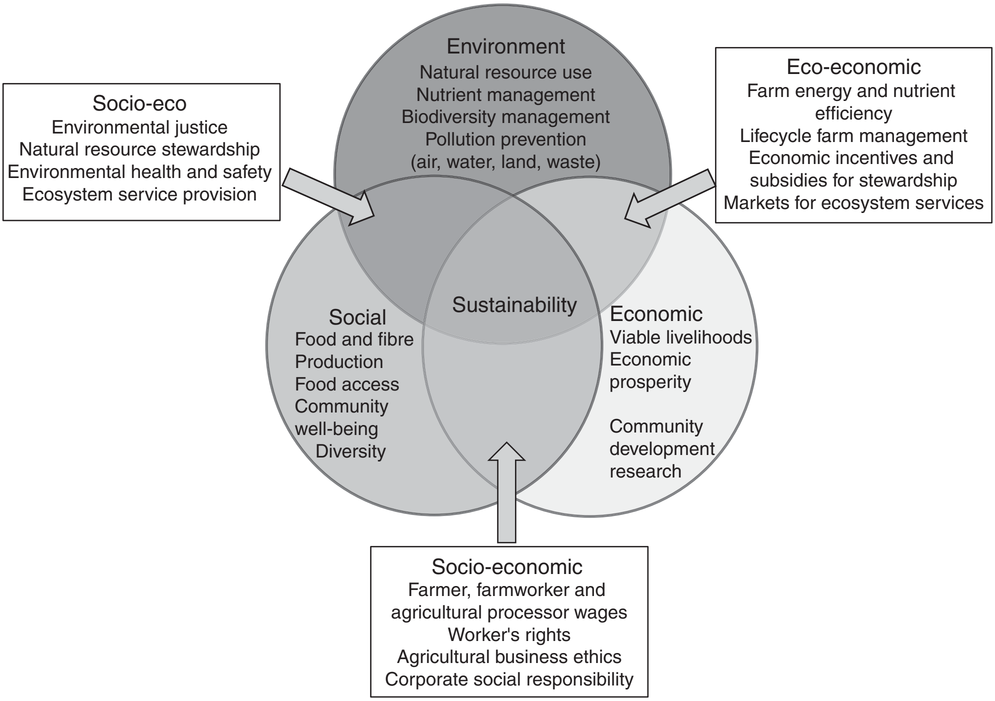


Fig. 13.1. The three pillars of sustainability – social, environment and economic – adapted to agricultural systems.

wage and had ethical labour practices (Browne *et al.*, 2000). While social sustainability has varying definitions, it can encompass many components of sustainability and naturally overlaps into the environmental and economic spheres including food security and access, community well-being, diversity and culture, environmental justice, living wages and economic farm viability.

Social Sustainability in Animal Agriculture

Agriculture and ranching provide significant social benefits to society through the provision of food and fibre, ecosystem services and community development. However, agriculture in the USA has changed significantly in the past several decades. Farmers, ranchers, farmworkers and agricultural processors are more diverse than ever before and agricultural markets are rapidly changing. At the same time, food security and access is less certain with globalization and a growing population and rural communities face new challenges for development. All of these changes are underscored by a rising population in developing nations like China and India amidst a global population expected to crest at 9 billion by 2050 (UNPD, 2009).

Challenges for achieving social sustainability in animal agriculture are significant particularly in a time of increasing global populations, rising demand for animal products and dwindling natural resources. Nevertheless, animal agriculture is also experiencing a new era of renewed interest in how and in what ways food is grown, which can provide potential benefits to farmers and ranchers. There are also increasing challenges and opportunities for animal agriculture from policy. New regulations can both provide potential economic benefits through eco-labelling schemes or payment for ecosystem services, and economic and social challenges as a result of immigration, environmental regulations and food safety. This section will discuss some aspects of social sustainability in animal agriculture including food access and security, farmer's markets, and diversity and culture.

Food access and food security (Farm Bill breakdown)

One of the key components of social sustainability in agriculture is the production of food and fibre. Farmers and ranchers provide significantly greater amounts of food today than in previous years, largely as a result of mechanization, traditional plant and animal breeding, and agricultural inputs such as manures and fertilizers. According to the United Nations Food and Agriculture Organization, there is enough food available in the world for each individual to consume nearly 2800 calories a day (UN FAO, 2007) – notably above the US Food and Drug Administration recommended daily 2000 calories a day (USFDA, 2004). Yet, despite continued agricultural growth and productivity, food security remains a significant problem globally. In 2010, it is estimated that there were 925 million undernourished people in the world (UN FAO, 2010). These statistics demonstrate that simply producing food is not enough to guarantee that people have access to adequate nutrition and the ability to acquire food directly. As a result, our world currently suffers from a food access problem, largely social and economic in nature, rather than an agricultural food production issue.

A sustainable food and agriculture system is one in which food is not only grown but is available and secure for all people. The 1996 World Food Summit defined food security as existing 'when all people at all times have access to sufficient, safe, nutritious food to maintain a healthy and active life.' Similar to sustainability, food security is based on three pillars including food availability, food access and food use (WHO, 2012). In the USA, an estimated 14.5% (17.2 million) of households were food insecure in 2010. Of these, 3.9 million households with children were food insecure (Coleman-Jensen *et al.*, 2011). These food insecurities in the USA are combated through a number of government programmes. In 2011, 46.2 million people (about one in seven Americans) participated in the government sponsored Supplemental Nutrition Assistance Program (SNAP) (USDA FNS, 2011a) with more than 9 million receiving assistance from the Women, Infants and Children (WIC) Program designed for pregnant and nursing women and children under the age of 5 (USDA FNS, 2011b).

Food assistance programmes in the USA are funded through the Farm Bill, a major piece of federal legislation that is renewed every 5 years. The Farm Bill covers multiple programmes through the US Department of Agriculture (USDA) including funding for farm and ranch subsidy programmes, farm and ranch insurance programmes and conservation initiatives. However, the majority of the Farm Bill goes to support 15 different food access and nutrition improvement programmes (Fig. 13.2). In the 2008 Farm Bill, 68% of funding was allocated to nutrition programmes including the National School Lunch Program, SNAP and WIC (Young *et al.*, 2008). These programmes provide SNAP and WIC recipients with access to fresh fruits and vegetables, dairy and cheeses, eggs, beans and groundnut butter. As well, the National School Lunch Program provides daily meals for more than 31 million American children (USDA ERS, 2011a).

More recent efforts have also been made in these programmes to incorporate aspects of environmental and economic sustainability. WIC and SNAP benefits can now be used at farmers' markets and farm stands throughout the country (Box 13.1). As well, the National School Lunch Program has initiated a variety of programmes to encourage local provision of agricultural products including grass-fed beef in some regions to help support local farmers and ranchers and reduce the distance food travels (Ulla, 2010). While the intent of government

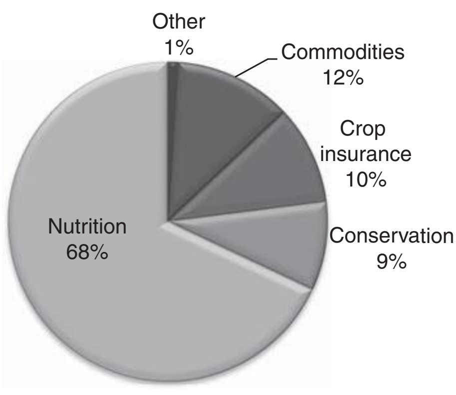


Fig. 13.2. Allocation of resources to programmes supported by the 2008 Farm Bill. Source: USDA Economic Research Service using Congressional Budget Office estimates.

Box 13.1. Combating food deserts through farmers' markets.

Despite programmes to increase food assistance in the USA and abroad many people globally suffer from malnutrition. The USDA recently launched a programme to identify 'food deserts' throughout the USA. Food deserts are defined as 'an area in the USA with limited access to affordable and nutritious food, particularly such an area composed of predominantly lower income neighborhoods and communities' (US Congress, 2008). Importantly, food deserts can afflict urban, suburban and rural regions. While urban regions may experience food deserts because of a proliferation of corner stores without adequate fresh fruits and vegetables, rural regions may also live considerable distances from grocery stores or markets. In fact, some rural areas that are largely comprised of farming and ranching land may have problems with food access. The USDA tracks food deserts through an online tool, which shows that significant portions of the agricultural regions of the Midwest are considered food deserts. Though many people in these regions may be farmers or ranchers themselves, it is possible that they lack access to affordable and nutritious food. The USDA estimates that of the more 6500 identified food desert regions in the USA about 25% are in rural areas (USDA ERS, 2009).

One promising way food deserts are being combated is through the rise in farmers' markets. According to the USDA, farmers' markets in the USA have risen 250% since 2000. As of 2011, there were 7175 farmers' markets throughout the USA in both urban and rural regions (USDA AMS, 2011). Farmers' markets are increasingly accepting vouchers and credits used in the SNAP and WIC programme, providing new choices for many Americans. In 2010, 18,245 farmers, 3647 farmers' markets and 2772 roadside stands were authorized to accept coupons from the Farmer's Market Nutrition Program in conjunction with WIC, providing more than US$15.7 million for farmers (USDA FNS, 2012). In 2009, SNAP benefits were accepted at 936 farmers' markets resulting in more than US$4.3 million in redemptions (USDA FNS, 2010). In addition to providing food access, these markets are helping agriculture meet additional sustainability goals by providing new markets for farmers and ranchers to ensure economic viability and increase environmental sustainability by reducing the number of miles a food travels to a consumer.

nutrition assistance programmes is foremost designed to provide food access and availability, when components of environmental and economic sustainability are considered, these programmes are able to provide benefits to agricultural communities and minimize environmental impacts.

Changing demographics and culture in animal agriculture

Social sustainability includes many components of diversity and culture. In the context of food, it can mean that individuals have access to food that is culturally appropriate among other things (Alkon and Norgaard, 2009). Socially sustainable food and agriculture should also consider the role of diversity and culture in production, processing and distribution. Agriculture in the USA has changed significantly in the past several decades as racial and ethnic diversity has increased. These changes in demographics and backgrounds have a fundamental influence on the agricultural industry as well as the types of products that American consumers demand.

Farming and ranching is changing in the USA. The Census of Agriculture estimates that between 2002 and 2007 there was a 10% increase in the number of principal farm operators of Hispanic origin. During this same time, female farm operators increased by almost 30% (USDA NASS, 2007). Agricultural workers are also increasingly of non-white backgrounds. It is estimated that though Latinos represent only 15.4% of the total US population, they account for more than 60% of all agricultural workers (Liu and Apollon, 2011). Workers in animal processing are also increasingly diverse. Between 1980 and 2000, the percentage of foreign-born workers in the meat processing industry climbed steadily. The percentage of foreign Hispanic workers increased from 49.6% in 1980 to 81.9% in 2000. During this same time frame, the percentage of whites working in the meat processing industry dropped from more than 70% to less than 50% while the number of Hispanic workers increased from less than 10% to nearly 30%. The percentage of Asian and Black meat

processing workers also increased slightly during this time. By 2008, the percentage of Hispanic meat processing workers hovered around 37% – slightly surpassing the total number of white workers (Kandel, 2009).

The changing population of farmers, agricultural workers and meat processors presents many potential challenges including language barriers, information access, immigration status and visas, and even marketing opportunities (Swisher *et al.*, 2007). Farm-workers and processors may not speak English and may not have adequate information about health and safety at the farm and factory level. They may also not have resources for understanding their legal rights. From the context of food access, changing demographics in agriculture also suggests that there are changing food needs to ensure that available food is culturally appropriate.

Unfortunately, though farm-workers and processors work with food, they are also at a high risk for being food insecure. In California, it is estimated that 45% of agricultural workers surveyed were food insecure with nearly half on food stamps (Wirth, 2007). In North Carolina, it is documented that nearly two-thirds of migrant and seasonal farm-workers are food insecure (Borre *et al.*, 2010). These rates are significantly higher than the average in the USA, making agricultural workers and processors a high-risk population for food insecurity. These demographics indicate that the agricultural industry is changing quickly and programmes and education designed for the agricultural community may also need to change. Cooperative Extension programmes are beginning to publish their materials in Spanish (Extension en Espanol, 2012) and many organizations now exist for women in agriculture. Recently, in conjunction with a lawsuit, the USDA established the Council for Native American Farming and Ranching to assist in USDA and Native American relations and resources. Many states also now offer information about their WIC and SNAP benefits in Spanish or other languages to cater to many races and cultures. Challenges still remain: the documented number of farm-workers lacking food security is cause for considerable concern. Ensuring that living wages are paid and that farm-workers have access to food and services continues to be a challenge throughout agricultural regions. This idea will be discussed further in the section on social–economic sustainability.

Social–Environmental Sustainability in Animal Agriculture

Social sustainability also encompasses other aspects of sustainability including an overlap with environmental issues. Social–environmental sustainability in animal agriculture involves issues related to both the environmental impacts of agriculture and the social components of food access, well-being and diversity. Issues related to social–environmental sustainability can provide both challenges and opportunities for producers, agricultural communities, the agricultural industry and policy makers. At the same time, social–environmental sustainability also considers the potentially unequal environmental impacts of agriculture on certain socio-economic groups or races of people.

Environmental justice

The US Environmental Protection Agency defines environmental justice as, 'the fair treatment and meaningful involvement of all people regardless of race, colour, national origin, or income with respect to the development, implementation, and enforcement of environmental laws, regulations, and policies.' The federal definition notes that environmental justice is comprised of two parts: fair treatment and meaningful involvement. Fair treatment ensures that no group of people is burdened by an unequal share of environmental or human health impacts because of business or government policies. Meaningful involvement means that individuals have the opportunity to participate and be considered in decision making that will affect their environment, health or both (CDC, 2011). In 1991, the First National People of Color Environmental Leadership Summit adopted 17 principles of environmental justice, which broadened the movement's focus to include all people, not only communities of colour. Three years later President Clinton signed an Executive Order (12898) on 'Federal Actions to Address Environmental Justice in Minority Populations and Low-Income Populations'. The order mandated that federal agencies develop strategies to manage environmental justice issues by identifying adverse programmes and policies that

disproportionately affect minority and low-income communities (Bowen and Wells, 2002).

Though there were laws in place as early as the 1964 Civil Rights Act to prevent environmental injustices, many communities have dealt with unequal distribution of environmental pollutants or been closed out of political processes. Historically, many environmental justice issues dealt with the zoning of toxic waste facilities and pollution exposure (Harvey, 1996). Studies have found that communities of colour or lower socio-economic status have greater exposure to a range of environmental pollutants including air pollution (Carson *et al.*, 1997) and hazardous waste (Mohai and Saha, 2007). Other studies have also found that this exposure or proximity to pollution is associated with increased health issues including cancer prevalence and mortality (Brulle and Pellow, 2006), though it is difficult to demonstrate that such exposures directly cause these health issues (Wakefield and Baxter, 2010).

Environmental justice concerns in agriculture

Environmental justice is an important topic to consider for animal agriculture, as agriculture can produce air and water pollutants. Greenhouse gas emissions as well as other toxins including hydrogen sulfide, particulates, ammonia and dust are potential concerns from animal facilities. Nutrients and inputs used in agriculture including synthetic fertilizers, manures and pesticides in excess can affect water systems and drinking water supplies. For communities located in agricultural regions, agriculture can provide significant economic benefits; however, agriculture can also contribute to environmental and human health concerns.

In some agricultural regions, particularly those that have concentrated animal feeding operations (CAFOs), residents have reported adverse health impacts including higher incidences of stomach and respiratory issues (Wing and Wolf, 2000; Merchant *et al.*, 2005; Mirabelli *et al.*, 2006). Odours are also a potential problem for residents living near livestock facilities, potentially affecting the health of communities as well as their quality of life (Thu, 2002). These pollution issues can be an environmental justice issue if they are concentrated in regions that disproportionately affect a group of people. Some research suggests that CAFOs are more frequently located in lower socio-economic and non-white communities (Durrenberger and Thu, 1996; Wing *et al.*, 2000; Mirabelli *et al.*, 2006).

One animal-producing region that has received significant attention for environmental justice concerns is North Carolina, where a large number of pig farms are located. In 2008, North Carolina was the second hog-producing state up from the 15th in 1985 (Wing *et al.*, 2000). At the same time, the number of small farms (fewer than 1000) dropped from more than 80% of total operations in 1993 to 40% in 2007 while the total number of operations over 4000 hogs rose from less than 10% to approximately 50% (National Hog Farmer, 2008). The concentration of hog farming into larger facilities was also followed by a concentration of facilities in the coastal plain region with characteristic low-lying flood plains and high water tables.

Studies have found that the location of these facilities throughout eastern North Carolina is not uniform; large-scale hog operations or CAFOs were disproportionately located in communities with higher poverty levels and higher proportions of non-white residents. In particular, the combination of high poverty and non-white resident populations resulted in the greatest concentrations of CAFOs. The characteristic liquid manure pools used in large-scale pig farming, in addition to the airborne emissions from these facilities, have created significant issues throughout the region. There is documentation of groundwater contamination from hog facilities in the region as well as a host of public health impacts (CDC NCEH, 1998). Following Hurricane Floyd in 1999, lagoons throughout the region burst or spilled their waste into local water systems, threatening public drinking water supplies and causing significant environmental damage (Kilborn, 1999).

However, while these impacts are disproportionately affecting low-income and non-white residents, the issue has been difficult to address in the community. The hog farmers themselves usually produce pigs on a contract through a large company, making on average slightly more than US$29,000 annually in 2006 (Employment Security Commission of North Carolina, 2006). This limits the ability of

individual farmers to address these issues. Instead, state policies and the industry have made attempts to minimize environmental waste. In 1997, a moratorium was placed on the construction of farms with more than 250 hogs or the expansion of existing hog farms in North Carolina. Several years later, Smithfield Foods, who operates the largest meat processing plant in the world in eastern North Carolina, developed an agreement with the Attorney General of North Carolina to fund research for environmentally superior technologies that could reduce potential environmental harm. Unfortunately, the project concluded that while there are several existing viable options for reducing environmental impacts, alternatives are too expensive for the average farmer (Collins, 2006). State policies have also been largely ineffective in dealing with environmental justice issues, as the state moratorium's loopholes allowed for more than 500,000 new hogs to be added on to farms between 1997 and 2007.

One future hope for the region is the emergence of carbon offset markets and payment for ecosystem services (discussed in further detail below). These programmes may offer some solutions for the future as hog farmers may be able to adopt practices such as anaerobic digesters and receive payment – thus reducing environmental impacts and improving their own economic viability (Subler, 2011).

Environmental health and safety in animal production

Along with potential health impacts in communities from animal facilities health impacts can also be a concern for farmers, ranchers and agricultural workers. Seasonal farm-worker communities may also not have access to food or clean water and may be disproportionately exposed to environmental contaminants (US EPA, 2012). For livestock facilities, one of the most dangerous air emissions is hydrogen sulfide, an acute gas that can cause death (Gerasimon *et al.*, 2007). Each year there are deaths from hydrogen sulfide poisoning as farmers and farm-workers enter manure pits or other confined spaces that are not adequately ventilated (NIOSH, 2007). Pesticide exposure is another potential hazard in agriculture. The Centers for Disease Control and Prevention estimates that annually in the USA there are between 10,000 and 20,000 pesticide poisonings. Farm-workers and their families are at a high risk for pesticide exposure, which can lead to health problems (Simcox *et al.*, 1995; Salvatore *et al.*, 2008).

Potential concerns for environmental health and safety for agricultural workers is also important beyond the ranch. Today there are nearly 400,000 people employed in meat processing throughout the nation (BOLS, 2011). More than 80% of meat-processing workers are people of colour with almost 60% of all meat-processing workers classified as Latino (Liu and Apollon, 2011). Meat processing jobs are consistently ranked as one of the most dangerous jobs in the USA as workers are highly susceptible to injury. The repetitive tasks of slicing and lifting can also lead to cumulative injuries like carpal tunnel syndrome and back strains (BOLS, 2011).

As the diversity of agriculture increases, it is important to consider environmental justice, and health and safety. Farm-workers need working conditions that prevent exposure to environmental and human health contaminants. This may in part be alleviated through increased language education for farm-workers, meat processors and employers to safeguard well-being. Education and outreach related to risk and exposure for hazardous working conditions or agricultural inputs can also assist in reducing potential farm-worker and processor exposures (Arcury *et al.*, 2002). Government and industry regulations can also play a significant role in reducing potential harm by increasing standards for safe working conditions.

Working to mitigate any environmental or public health impacts resulting from agriculture is challenging since it is often difficult to quantify and demonstrate impacts. Furthermore, ensuring that claims about environmental pollution are substantiated can also be difficult. Non-farmers or ranchers may submit environmental complaints for practices that may be accepted in agriculture. For example, one study found that citizen environmental complaints of livestock facilities were overwhelmingly for odour and surface water; however, the odour complaints were often for accepted management practices in animal agriculture that did not

require mediating action according to current law (Hadrich and Wolf, 2010). This highlights the need for community engagement between farmers, ranchers and those not working in agriculture to discuss appropriate agricultural practices, environmental health and safety measures.

Payment for ecosystem services

Pasture and rangelands are located in all 50 states in the USA with privately owned pasture and rangelands making up more than 27% of the total hectarage of the lower 48 states. As a result, ranchers can be an important asset for private land management and the provision of ecosystem and social services. Ecosystem services are defined as 'the benefits people obtain from ecosystems'. These can include provisioning services (i.e. food), regulating services (i.e. flood regulation), supporting services (i.e. nutrient cycling) and cultural services (i.e. recreational) (Millennium Ecosystem Assessment, 2005). Payment to the agricultural community for ecosystem services can supply new opportunities for farmer and rancher income while providing social and ecosystem benefits that are highly valued (Kulshreshtha *et al.*, 2012). Pasture and rangelands provide recreation opportunities, scenic, cultural and historic values that improve social well-being and quality of life (USDA NRCS, 2011). In many ways, grazing lands and ranching can also provide significant environmental benefits to the soil, biodiversity and wildlife, and even reductions in global greenhouse gas emissions.

Managed grazing can provide both benefits and challenges to wildlife. Agriculture and pasture conversion can be a significant cause of habitat loss for numerous species (Javorek *et al.*, 2007). However, when well managed, agriculture and grazing lands can also provide biodiversity benefits. Grazing can be beneficial for biodiversity by minimizing non-native grasses in an ecosystem and promoting habitat for multiple species. Forbs, a native species of concern in the Western USA, can have a higher abundance in grazing lands than non-grazed lands (Hayes and Holl, 2003). Grazing can also have a significant impact on wildlife. By reducing exotic species and their density, grazing can provide beneficial habitat for a number of endangered species including kangaroo rats, squirrels and leopard lizards (Germano *et al.*, 2001). The checkerspot butterfly is another species that can benefit from grazing livestock systems. Evidence suggests that in systems where grazing stopped the butterfly populations crashed, but were maintained in systems with continued grazing (Weiss, 1999). Grazing can also contribute positively to the preservation of vernal pools, seasonal ponds that develop in some regions and are a breeding ground for the endangered tiger salamander (Pyke and Marty, 2005).

Managed livestock systems can also provide significant social–environmental benefits through carbon management. Globally, grasslands contain 20–25% of global terrestrial carbon (Kimble *et al.*, 2001). In the USA, there are 336 mha of grazing lands, 48% of which are in rangelands. Grazing lands can provide carbon storage in their soils, which can reduce global greenhouse gas emissions. As a result, preserving grazing lands and maintaining well managed grazing systems can be a strategy for providing the public with reduced greenhouse gas emissions (Schuman *et al.*, 2002).

As our knowledge of grazing systems increases and more information has been made available about the benefits of grazing practices, policies to encourage the sustainable management of grazing systems have increased. Payment for ecosystem services involves providing incentives to farmers and ranchers to manage their land in a way that provides environmental benefits (and in turn social benefits through land conservation, reduced pollutants and increased biodiversity). Ecosystem services payment can contribute significantly to sustainability goals of incorporating the economic, social and environmental components. Farmers and ranchers can receive an economic incentive, while the environment can be improved for the benefit of society and wildlife.

There are many types of ecosystem service payment programmes. Some ecosystem service programmes are run through government programmes including the US-based Natural Resources Conservation Service (NRCS) and the Environmental Quality Incentives Program (EQIP) implemented through the USDA. NRSC

programmes run through USDA include payment for grassland, farm, ranch and wetland preservation or restoration. EQIP funding can provide farmers and ranchers with payments to help share the cost of implementing practices like renewable energy, energy conservation, nutrient management and water enhancement. More recent programmes have focused on a market-based approach where producers who implement practices to reduce water pollution can receive credits that can be sold in a market. In 2012, the Secretary of Agriculture announced a new water quality trading programme in the Chesapeake Bay region, where poultry farmers in particular can take advantage of such a programme (USDA, 2012). Additional programmes are continuing to emerge and provide opportunities to increase sustainability in grazing systems.

Carbon offset markets for sustainability

Carbon trading markets are another ecosystem service payment programme increasing in the USA and globally, some of which allow for farmers and ranchers to implement practices to reduce greenhouse gas emissions and trade their permits in a market. While carbon markets provide a tangible economic benefit for farmers and ranchers, they also provide ecosystem services and social benefits through reduced greenhouse gas emissions. The Climate Action Reserve, a non-profit organization developing a series of protocols for agricultural greenhouse gas emission reductions, first adopted a US livestock protocol for methane digesters in 2007 (Climate Action Reserve, 2010a). Through this programme, farmers can adopt a methane gas digester to reduce methane emissions and sell credits for those emissions in a carbon market. A similar programme was developed for livestock in Mexico in 2009 (Climate Action Reserve, 2010b).

Government programmes are also advancing carbon markets and trading that include agriculture. In Alberta, several protocols have been developed for reducing greenhouse gas emissions in livestock, dairy and pork production (Government of Alberta, 2011a). For livestock producers, eligible practices include feeding edible oils, reducing the age of livestock at harvest, or reducing the number of days on feed (Government of Alberta, 2011b). Farmers and ranchers implementing these practices can receive credit for their new practices and sell them in the market to earn additional income. In this context, carbon markets can be promising for sustainability by fulfilling all three components of sustainability – reducing environmental impact, social benefit from food production and reduced public health impacts from lower emissions as well as additional income for farmers and ranchers to maintain their operations.

Livestock production in Central America is an important livelihood for many farmers; however, deforestation continues to be a major problem associated with agriculture. For example, in Nicaragua, it is estimated than 26% of the nation's native forest was cut down between 1995 and 2000. To combat this deforestation and encourage the development of silvopastoral systems (management that plants trees or shrubs into grazed, permanent pastures) payment for ecosystem services have been implemented by governments, the United Nations Food and Agriculture Organization (FAO) and the World Bank. Ranchers are able to receive payment if they adopt a number of different types of practices including planting trees, shrubs or live fences around cattle pastures (Fig. 13.3). These practices can provide significant social and environmental benefits by increasing carbon storage and biodiversity, while maintaining food production (Garbach *et al.*, 2012). Though payments are modest so far (about US$2 a day in the FAO/World Bank programme), the programme appears to be working. In Costa Rica, participating farmers reduced degraded pasture by more than 60%, storing an estimated 25,000 t of carbon. Ranchers have also seen significant increases to biodiversity as a result of these policies with an increase in bird species on farms adopting ecosystem service practices compared with those not implementing new practices (UN FAO, 2006). Farmers have stated that the extra income associated with payment for ecosystem services is a significant motivating factor behind the adoption of these practices for sustainability (Garbach *et al.*, 2012).

Fig. 13.3. M. Sotelo measures characteristics of *Erythrina* trees in a recently planted live fence on farmlands in Turrialba Costa Rica (Photo: Courtesy of K. Garbach 2010).

Socio-economic Sustainability in Animal Agriculture

As previously mentioned, payment for ecosystem services can provide both social benefits as well as economic benefits to farmers and ranchers. Ensuring the long-term sustainability of animal agriculture requires farm and ranch economic viability to prevent farmland from being lost to development. However, socio-economic sustainability also includes economic viability for farm-workers, ranch hands and meat processors. Though certification programmes have emerged particularly in developing countries to ensure that fair wages are paid to farmers, there are few initiatives for fair trade in developed nations.

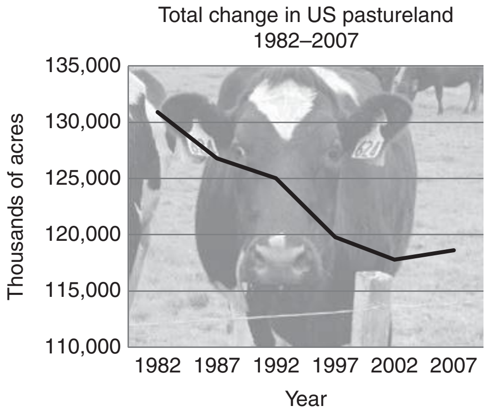
Agricultural land loss and economic viability

Economic sustainability in agriculture is often overlooked as one of the central tenants of sustainable agriculture. As mentioned in the introduction to this chapter, there are plenty of eco-labels in the market place to help consumers choose between varying levels of environmental sustainability. Throughout the 2000s, consumers became increasingly interested in the way in

which their food was grown and the environmental footprint of different food products. Yet, the economic viability of farming and ranching is also a crucial component of sustainability – without farms and ranches society loses significant benefits in the form of agricultural production, ecosystem service provisions and jobs.

The loss of farm and ranchland to development is a major threat to all aspects of sustainable agriculture. Between 1982 and 2007 more than 23 million acres of agricultural land were developed – an area the size of Indiana (American Farmland Trust, 2011). Total US pastureland decreased by more than 9% in the same time frame, while rangeland decreased by more than 2% (USDA, 2009a) (Fig. 13.4). Farm loss itself was even more significant – in just the 5 years between 2002 and 2007, the total number of cattle farms/ranches decreased by 5% (USDA, 2008a) while the total number of dairy farms dropped by 21% in the USA. Unfortunately, in this same period, total production expenses increased by 35% in dairy and 30% in cattle ranching. Two states with significant contributions to the animal agriculture sector – Texas and California – lost the most amount of agricultural land during this period, respectively losing 2.9 million and 1.7 million acres (American Farmland Trust, 2011). In 2007 Texas accounted for the greatest total value of US sales for cattle and calves while California accounted for the greatest amount of dairy (USDA 2008a, b). As the cost of farming and ranching has increased, this has had a direct impact on the decreasing amount of farm, pasture and rangeland.

As agricultural land continues to disappear farmers and ranchers face increasing costs and continue to receive a smaller proportion of the total money consumers spend on purchasing food. The farm share is a term used to describe the amount of every dollar spent on food that is actually received by farmers. Since the USDA began measuring farm share in 1993, farmers and ranchers have consistently seen their farm share decrease. In 1993, farmers received 18.4 cents for every US$1 spent on food in the USA. In 2008, this number had dropped to only 15.8 cents for every dollar. The greater than 80% left of the food dollar goes to the marketing share, which is the amount of money given to food supply chain industries involved in post-farm activities like processing, transportation and marketing. While the loss of agricultural land can be attributed to many issues beyond simply economic terms, rising costs in animal production along with the decreasing share of the average food dollar spent for farmers and ranchers is certainly one important aspect (USDA ERS, 2011b).



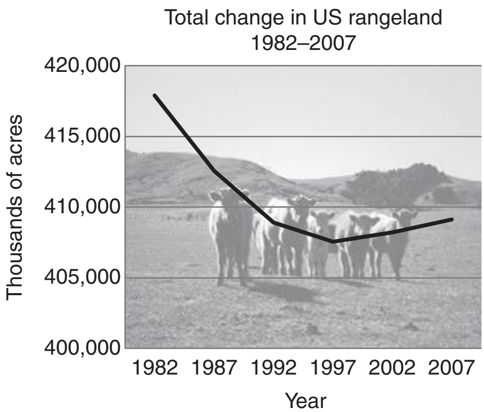


Fig. 13.4. Loss of pasture and rangeland in the USA between 1982 and 2007. While recent years have shown a slight increase in total pasture and rangelands, overall there is a net loss. The loss of agricultural land is a direct threat to the sustainability of agriculture and food production. Source: National Resources Inventory, USDA NRCS and Iowa State University, 2009.

Combating agricultural land loss with conservation easements

Fortunately, there has been an effort to combat the loss of agricultural land in recent years.

In addition to payment for ecosystems services designed to provide additional income to farmers, there is also a rise in conservation easements and land trusts to preserve agricultural land. Conservation or farm easements are a way for landowners to protect their land and preserve open space. An easement on a farm or ranch limits the ability of the land to be developed or subdivided, but maintains ownership with the individual who applied for the easement. Conservation easements can be implemented by government and non-government organizations (land trusts), which upon acceptance of the easement agree to monitor it forever to ensure compliance (Maryland Department of Natural Resources, 2012). In the USA, there are more than 1700 land trusts that have so far conserved 37 million acres, roughly the size of New England. One reason for the curtailing of pasture and rangeland loss in the USA may be the rise in conservation easements and land trusts, which can be an effective way for private lands to be protected for sustainability (Land Trust Alliance, 2012).

Fair wages and fair trade in agriculture

As the farm share has dropped steadily over the past two decades, so has the percentage of farm costs spent on labour and benefits. In 1997, nearly 4% of total costs were spent on salaries and benefits, while in 2008 this amount was 2.86% (USDA ERS, 2011b). In some cases, this may because of the increase of mechanization in agriculture, which has likely played a role in the decrease in agricultural workers in the USA. Between 1990 and 2010 hired farm-workers declined from more than 1.14 million to 1.05 million, with an estimated 38% of hired farm-workers employed in the livestock industry, mostly on larger farms and ranches with sales more than US$500,000 a year. During the same time period, farm-worker wages rose by about 0.9% compared with 0.6% for non-farm non-supervisory positions (USDA ERS, 2011c). Nevertheless, the average wage for a farm-worker in 2010 was US$10.17 an hour and an estimated 30% of all farm-workers had family incomes below the poverty level in the early 2000s (US DOL, 2002).

Farm-worker wages in food production are also not equally distributed. According to data from a 3-year survey among food industry workers, people of colour consistently make less in food and agriculture production jobs. For example, among agricultural workers (including jobs like animal breeding, delivering animals, planting and harvesting crops) people of colour represent more than 65% of the work force, but earn on average 22% less per year than white individuals. For agricultural inspectors, people of colour, also representing about 65% of the work force, made nearly half as much as their white counterparts. Similar wage discrepancies can be found among poultry, meat and fish processing workers where more than 80% are people of colour, making US$2500 less per year on average than white individuals in similar positions (US$23,645 for white workers compared with US$21,117 for people of colour) (Liu and Apollon, 2011).

One solution to wage discrepancies has been the rise in the use of fair-trade labels. Fair trade is the idea that producers can receive better payment for their product as certified by intermediary organizations like Fairtrade International. In practice, there are a host of products available to be certified fair trade ranging from bananas to coffee to cotton to sports balls. These programmes allow consumers usually in Western countries to purchase products from developing countries at a premium for ethical working conditions and wages (Fairtrade International, 2011). In western nations like the USA, there have not been similar programmes until recently when a domestic North America-based fair trade association began to ensure fair wages and treatment of workers in the USA (Domestic Fairtrade Association, 2010). Despite this, there are currently no fair trade standards in place for meat, dairy, or other animal products. And while fair trade products may provide additional income for developing world and potentially developed world farmers and farm-workers, they are also more expensive than their conventional counterparts, often making them infeasible to consumers who cannot afford to pay extra for products.

Moving Forward – Reconciling Multiple Sustainability Goals

Animal agriculture has reached a new era of sustainability in which more opportunities are available to ensure that animal products were raised in a way that was socially, economically and environmentally sustainable. Trends in consumer demands suggest that some people are willing to pay more money for products that can certify that they have taken steps to be more sustainable. Yet while some consumers are able to afford what are sometimes more expensive food costs for such products, other data show that there are increasing numbers who are food insecure and relying on food assistance. Farmers and ranchers also face threats to sustainability from economic impacts like loss of farmland to development and decreasing farm shares over time. In addition to rising costs of production, farmers are often unable to implement sustainable practices because of their added cost in labour, time or resources.

Sustainability goals of multiple stakeholders

This potential discrepancy related to sustainability is complex as often the three pillars of sustainability are at odds with themselves. For example, a dairy farmer may wish to reduce their overall greenhouse gas emissions and odour from manures by implementing a manure methane digester. A digester can also provide economic benefits to the farmer because they can co-generate electricity through the process that can be used on their farm. However, the average upfront capital cost of a digester on a dairy farm can range from nearly US$500,000 to more than US$1 million (Giesy *et al.*, 2009). For most farmers, this cost puts the option out of the range of possibility. Similarly, efforts to raise wages in the agricultural industry may not be possible if entirely covered by farmers whose farm share continues to drop.

Recent research also demonstrates that farmers' concerns related to climate change may be fundamentally different than those of policy makers, the public, or environmentalists. Jackson *et al.* (2011) and Niles *et al.* (2013) found that farmers' greatest future concerns related to climate change were for regulations, fuel and energy costs, and more volatile markets – not the environmental impacts anticipated from changes in temperature and water. In contrast, policy makers and environmentalists have mostly focused on agricultural sustainability related to climate change in terms of the greenhouse gas emissions created by agriculture or the potential climate change impacts to agriculture that may reduce yield. This mismatch in sustainability goals or concerns suggests that there are multiple goals to be achieved in animal agriculture sustainability by different stakeholders.

The role of policy in sustainable animal agriculture

Despite these challenges, policies and regulations do offer the potential to help blend goals of sustainability in animal agriculture. Government regulations such as the establishment of a minimum wage have helped to raise standards of living for all workers including agricultural and food industry processors, though others within the agricultural industry likely experienced a loss in profits as a result of these policies. Environmental regulations and policies are also striving to increase agricultural sustainability, often in concert with agricultural producers. In 2009, USDA announced an initiative with the dairy industry to provide funding to establish anaerobic digesters and energy audits across the USA. Through government funding, 30 farms were able to implement digesters thus reducing the overall environmental impact of greenhouse gas emissions as well as the societal impacts of odours in rural communities (USDA, 2009b). One challenge of implementing voluntary agreements such as this one, is trying to ensure that actual gains in sustainability are achieved. Other policies that are mandatory such as regulations or performance standards (i.e. requiring a certain type of engine to be used on tractors) can help to guarantee that there are tangible environmental benefits achieved. Including agricultural producers, farm and ranch workers, food processors and other stakeholders in the policy process can assist policy makers in creating standards and

agreements that can match agriculture's own sustainability goals and achieve greater environmental and social sustainability in animal agriculture without compromising economic sustainability.

Consumer trends and demands

Consumer demand also continues to drive trends in animal agriculture, particularly as some consumers choose products like grass-fed beef, organic dairy and free-range eggs. There may be additional opportunities in labelling and consumer outreach to assist in guiding other sustainability goals in social and economic spheres. Labels and initiatives to safeguard agricultural land and individual farms may gain traction among consumers that wish to prioritize economic sustainability. Fair trade labels and other social sustainability labels are growing as new products are certified and new initiatives enter into western markets. As long as there is a demand for certain types of products, it is likely that animal agriculture production can respond to these changes and tap into markets that may assist farmers and ranchers in achieving their own goals for sustainability.

Conclusion

Sustainability in animal agriculture has never been more important as our world faces a growing population, limited natural resources and changing social challenges. Social sustainability is a key component of any sustainability initiative and must be considered in animal agriculture. In the quest for sustainability, it is also necessary to consider all people that work in the agricultural industry including farmers, ranchers, farm and ranch workers, and agricultural processors. Policies and consumer demand can continue to drive changes in agriculture, though it is important to recognize that there are multiple goals of sustainability from many stakeholder perspectives. In some cases, one type of sustainability may be at odds with another. Achieving sustainability in animal agriculture thus must consider all three pillars of sustainability and engage with the many groups working to achieve sustainability in the future.

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14 Life Cycle Assessment in Ruminant Production

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Introduction

Continued global expansion of livestock production to meet the demands of an increasing world population has prompted concerns over potential environmental impacts. Of primary concern are the use of resources (such as land, water and fossil fuel) needed to produce animal products, and the resulting excretions and emissions arising from animal production (Pullar *et al.*, 2011; see Chapters 2–5, this volume). Life cycle assessment (LCA) offers a means to quantify the environmental impact of livestock production, and to identify mitigation options that can be implemented to lower the environmental burden. This chapter presents the basic concepts of LCA and examines how LCA can be used to assess the environmental impact of livestock production. Our review focuses on ruminant production systems and global warming potential with limited reference to non-ruminants, as ruminant livestock have been identified as a greater contributor to greenhouse gas (GHG) emissions (Steinfeld *et al.*, 2006). Further details on the use of LCA for pig (Hermansen and Kristensen, 2011; Lammers, 2011) and poultry (Pelletier, 2008; Hermansen and Kristensen, 2011) production are available elsewhere.

Life Cycle Assessment in Animal Production

LCA is a holistic tool used by many industries to quantify the environmental impact of individual products (Ekvall and Weidema, 2004). The use of LCA is often prompted by the demand for accountability by customers, stakeholders and government regulators. Increasingly, industry uses results from LCAs to promote green-purchasing programmes whereby products are promoted based on their environmental performance. In such schemes, retail products are labelled as ecofriendly specifying the carbon (C) footprint or other environmental assessments (e.g. Environmental Product Declaration; (www.environdec.com). The intent of such labelling is to declare a particular product more environmentally sound than another, or to compare new alternative ecofriendly products with existing ones.

The International Organization for Standardization (ISO, 2006) defines LCA as the compilation and evaluation of inputs, outputs and potential environmental impact of a product system throughout its life cycle. An LCA that includes raw material acquisition through to production, use and disposal is referred to as a 'cradle-to-grave' analysis. Many LCAs have been

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conducted for industrial materials and manufactured products, and recent years have seen a rapid expansion of LCAs being conducted in agricultural production. There is rapidly expanding interest in LCA by the livestock industries, particularly by the beef and dairy sectors.

In animal agriculture, a partial LCA is often used to examine the product system up to the farm-gate because with meat, milk and eggs, most (>70%) of the emissions are from farm-related activities (inputs and on-farm activities), rather than the post-farm-gate activities of manufacturing, retailing and consumption. As farm activities are usually independent of downstream activities beyond the farm gate, changes in farm management to reduce environmental impact can carry through to the final product. In contrast, partial LCAs are usually avoided for manufactured goods because of the unintended potential of shifting the environmental burden to another part of the system. However, for most livestock products, a farm-gate LCA does not 'shift the problem' to another stage of the product's life cycle because in most cases farming activities are completely separate from manufacturing and retailing of the final product.

While LCAs may be useful for comparing the environmental performance of manufactured products, these comparisons can be very misleading for livestock products. The complexity of livestock farming systems, combined with the differences in methodologies of estimating the environmental impact of the farming system across studies makes it difficult to compare environmental performance of livestock production across animal production systems used worldwide.

One of the most useful outcomes of conducting an LCA for animal production is to identify opportunities for improving environmental performance within a production system. An LCA can help identify components or processes within the production cycle that should be targeted to reduce environmental impact. For example, Beauchemin *et al.* (2010) conducted a farm-level LCA of GHG emissions from beef production in western Canada. The LCA indicated that the largest contributing GHG in beef production was enteric methane (CH_4) accounting for 63% of total emissions, with about 84% of the enteric CH_4 from the cow–calf herd, mostly from mature cows. The LCA led the authors to conclude that research into mitigation practices to reduce GHG emissions from beef production should focus on reducing enteric CH_4 production from mature beef cows. It was thus shown that a proportional reduction in emissions from the cow–calf system would reduce whole-farm emissions by about four times as much as the same reduction in the feedlot system.

LCA can help decision making in terms of setting priorities for research and policy, as well as help prioritize mitigation options. For example, a change in the animal's diet might be recommended as a means of reducing enteric CH_4, but the change in diet may also lead to increased nitrous oxide (N_2O) from fertilizers used to produce the feed. A particular farming practice may appear to be a valid means of reducing GHG emissions from livestock production, but the complex interactions and feedbacks among farming practices can make it difficult to assess the net impact of the change. An LCA approach can be used to integrate the entire system, accounting for all changes in GHG emissions and removals arising from a prospective mitigation practice, such that the true impact of a change in practice can be fully evaluated (Janzen *et al.*, 2006).

Components of Life Cycle Assessment

LCA is conducted according to published international standards (14040 series; Life Cycle Assessments) developed by the ISO in 2006. The ISO is a private organization that aims to standardize and integrate quality aspects into business practices worldwide (Guinée *et al.*, 2002). The ISO (2006) standards can be used to determine the potential environmental impact of manufactured and agricultural products. Thus, the basic methodology is common to all products; however, the goal, scope, systems boundary and functional unit (FU) considered within the LCA can differ.

We present the main characteristics of LCA relevant to conducting farm-level LCA in this chapter, but readers seeking detailed methodological information on LCA should consult the ISO 14040–14043 standards (ISO, 2006) and the *Handbook on Life Cycle Assessment – Operational Guide to the ISO*

Standards (Guinée *et al.*, 2002). Readers interested specifically in using LCA for estimating GHG emissions are referred to the specification of the British Standards Institute (BSI, 2011a, b), which builds on the ISO standards to provide a method for assessing the life cycle GHG emissions of goods and services.

There are four phases of a LCA:

1. *Goal and scope*: The goal and intended use of the LCA set the depth and breadth needed in the study. These considerations will affect the system boundary (what elements are to be included) and the level of detail needed.
2. *Inventory analysis*: The inventory phase includes the collection of relevant input and output data.
3. *Impact assessment*: The impact assessment provides an understanding of the environmental significance of the results. This is done using indicators to reflect the environmental issue or impact category of interest (e.g. kg CO_2 equivalent (CO_2e) for GHG emissions).
4. *Interpretation*: The final stage leads to conclusions and recommendations based on the original goal of the LCA.

Types of Life Cycle Assessment

There are two basic types of LCA – attributional and consequential. An attributional LCA analyses the environmentally relevant flows to and from the system, whereas a consequential LCA describes how the environmental flows within a system might change in response to a change in product output (Finnveden *et al.*, 2009). For example, an attributional LCA might quantify the environmental impact of a dairy production system, while a consequential LCA might assess the future environmental burden of that same dairy system assuming higher milk production (Thomassen *et al.*, 2008a). To date, most LCAs in animal production have been attributional mainly because consequential LCAs are conceptually complex and are required to be dynamic. Consequential LCAs rely heavily on assumptions that directly affect the outcome and often there is a lack of basic information available. However, as LCAs are increasingly used to direct agricultural policy, there is a growing need for consequential LCAs to examine animal production under future scenarios.

Functional Unit

The Functional unit (FU) is the measure of output from the system and provides a reference point for expressing the environmental impacts. For animal production systems, the FU is typically a kilogram of weight (either live weight leaving the farm, shrunk weight at the abattoir, or carcass weight) for beef and sheep, and a kilogram of milk for dairy production. Milk is usually normalized as energy corrected milk (ECM) or fat and protein corrected milk (FPCM) to account for differences in milk components (note, ECM and FPCM are virtually interchangeable, differing by 1%) (Tyrrell and Reid, 1965). The International Dairy Federation (IDF, 2010) recently published guidelines for conducting LCA for the dairy industry and recommended the use of FPCM as the FU, with fat corrected to 4% and protein corrected to 3.3%. It should be noted that the IDF (2010) equation predicts slightly lower FPCM at a given milk composition compared with most other equations (Tyrrell and Reid, 1965).

When the goal is to determine efficient use of land resources, a more appropriate FU may be land use area (e.g. Beukes *et al.*, 2010; Foley *et al.*, 2011). In this approach, land use area includes the amount of land used directly by the farming operation as well as the amount of land required to produce any feed inputs. Land use can be a particularly relevant FU for assessing the impact of grazing management systems, such as stocking rate. In addition, land use has been used as a means of comparing resource usage across the various livestock commodities. For example, de Vries and de Boer (2010) compared 16 LCA studies from OECD countries using a FU of land use per amount of average daily intake of animal product. The daily consumption of beef (60 g day^{-1}) in OECD countries had the highest land use (1.65–2.96 m^2, followed by consumption (545 g day^{-1}) of milk (0.62–1.1 m^2), pork (82 g day^{-1}, 0.73–0.99 m^2) and chicken (74 g day^{-1}, 0.60–0.73 m^2), which were relatively similar, and then consumption of eggs (36 g day^{-1}) with the lowest land use value (0.16–0.22 m^2). The large amount of land required for beef production can be attributed to a low efficiency of converting ingested energy into edible meat coupled with the relatively small number of progeny produced per cow annually. However, it must also be stressed that beef cattle

can be raised on pastureland that would not support other forms of agriculture, thus a FU that considers type of land may be a more appropriate means of comparing amongst livestock commodities. For example, Wilkinson (2011) proposed that a more appropriate comparison between livestock systems would account for the proportions of human-edible and inedible feeds.

As the essential component of the human diet provided by meat, milk and eggs is protein, some have argued that a more appropriate FU for comparing livestock products is amount of protein or energy provided (de Vries and de Boer, 2010; Dyer *et al.*, 2010). In an analysis of GHG emissions from livestock production using a FU of kg CO_2e kg^{-1} of protein, de Vries and de Boer (2010) reported for OECD countries that production of milk (24–38), pork (21–53), chicken (18–36) and eggs (30–38) were relatively similar, but beef was much higher (75–170). In a Canadian study, Dyer *et al.* (2010) reported kg CO_2e kg^{-1} of protein as 119 for beef, 32 for milk, 25 for pork, 22 for eggs and 11 for chicken. These studies indicate that beef production is a relatively inefficient means of producing protein from the perspective of GHG emissions, a finding that needs to be considered relative to the many ancillary environmental benefits of beef production. Grazing and forage lands used in beef cattle production play an important role in preserving and building soil carbon reserves, conservation of biodiversity, water quality and wildlife habitat (Janzen, 2011). Thus, choice of FU for the LCA is an important consideration and will greatly affect the outcome and interpretation.

System Boundary

The system boundary defines the processes to be included in the LCA. The choice of elements included in the assessment depends on the goal and scope of the study (ISO, 2006). In undertaking a LCA of livestock products the system boundary might include the production and transport of inputs used on the farm, livestock production on the farm, transportation of products from the farm and between sectors, processing, wholesaling and retail distribution of the product, and finally handling and consumption by the consumer (Fig. 14.1). Inputs to the farm may include feed, feed additives, implants, fertilizer, herbicides, manufacture of tractors, barns and other capital equipment etc. but how far upstream the inputs are traced is a difficult issue, and is often hampered by lack of information or is associated with high

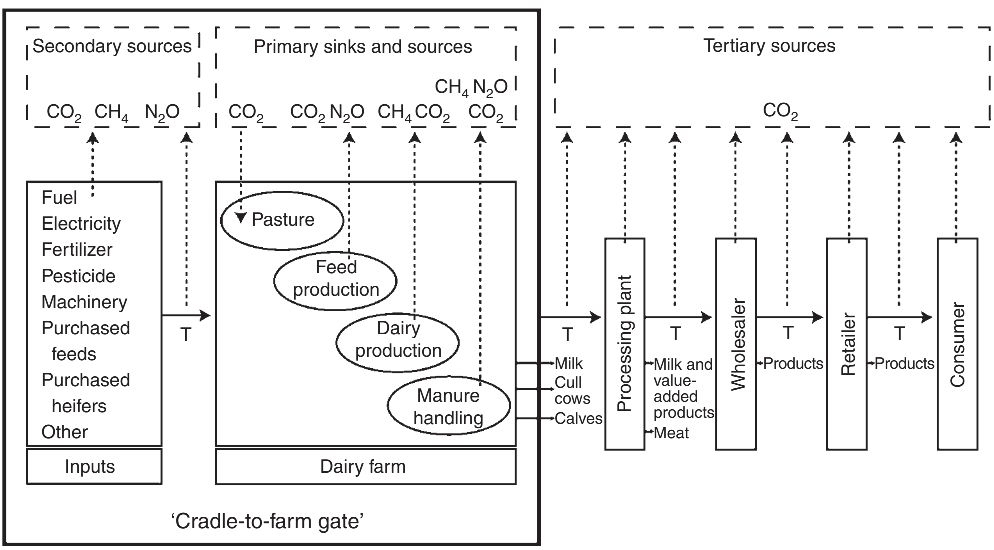

Fig. 14.1. System diagram showing the main life cycle stages, including inputs, outputs and flows, in a theoretical dairy production system as well as the system boundary for a 'cradle to farm gate' life cycle assessment (adapted from Rotz *et al.*, 2010 and Ledgard *et al.*, 2011; T, transportation).

uncertainty. Lack of consistency in the system boundary used in various LCAs of livestock products makes it difficult to compare across studies and makes interpretation of results uncertain.

As the largest proportion of the GHG emissions associated with producing meat and milk occurs up to the product leaving the farm, most LCAs of livestock products use a system boundary that ends at the farm gate, i.e. 'cradle to farm-gate'. For example, a LCA for lamb produced in New Zealand and consumed in the UK reported that 80% of the GHG emissions were from the cradle to farm-gate stage with only 20% from manufacturing, retailing and transportation activities beyond the farm gate (Ledgard *et al.*, 2011). Similarly, in a study of global dairy production, it was estimated that cradle to farm-gate emissions contribute on average 93% of total GHG emissions (Gerber *et al.*, 2010; 73–83% in developed countries). Because the greatest proportion of the GHG emissions is associated with the cradle to farm-gate stage, it can be advantageous to produce livestock in areas with a natural resource advantage and then transport the final product to markets in less advantaged areas. In contrast, grains for human consumption tend to be more efficiently produced close to the intended market because transportation and activities beyond the farm gate represent a much greater proportion of the total C-footprint.

Co-product Allocation

Livestock production systems can, in some cases, generate more than one product, which raises the issue of how environmental burdens should be allocated across the co-products. For example, dairying produces milk as the primary product, but substantial quantities of meat are also produced from cull animals and surplus calves. To deal with this complex issue, the ISO (2006) standard recommends that whenever possible, allocation of environmental burden be avoided meaning that environmental impact should be assessed for each co-product separately. That is rarely possible with livestock production systems because the production of the co-products is mutually dependent. In that case, several options are available for dealing with allocation of the environmental burden to co-products. These include: (i) system expansion, wherein the environmental burden is assigned to the co-product based on a parallel production system (e.g. for a dairy–beef system, the burden would be assigned to the meat based on a parallel traditional beef production system, which assumes that meat from dairy production displaces an equal amount of beef production); (ii) allocation on the basis of a physical relationship between the products (sometimes also called biological allocation); and (iii) allocation on the basis of economic value. There is considerable disagreement amongst experts on the best way of handling allocation, with valid arguments in favour of and against each method.

System expansion is often considered the best approach (Weidema, 2001; Cederberg and Stadig, 2003; Weidema and Schmidt, 2010), but this implies that there is an alternative means of producing the product and that the environmental burden of that production system is known. System expansion generally results in the lowest GHG intensity of milk production, compared with other methods of allocation (Flysjö *et al.*, 2011a). However, system expansion can be difficult to apply because LCA for substitute products may not exist or an alternative production system may not exist. Thus, in most LCAs of livestock production, both economic and physical allocation has been applied. With economic allocation, the environmental burden is allocated to co-products based on their respective price, usually averaged over some period of time (e.g. Kristensen *et al.*, 2011; McGeough *et al.*, 2012). A physical allocation can be applied by partitioning the feed energy consumed to the requirements for milk and meat production. For example, a LCA of Swedish milk production used physical allocation based on energy partitioning, which resulted in an 85% allocation to milk and 15% to meat (Cederberg and Stadig, 2003). The International Dairy Federation (IDF, 2010) also recommends a physical allocation of environmental burden to milk and meat, as follows:

$$\text{Allocation factor (AF)} = 1 - 5.7717 \times R,$$

where R = Mmeat/Mmilk, and Mmeat is the sum of live weight of all animals sold (including calves and culled cows) and Mmilk is the sum of milk sold (corrected for true protein and fat content). A default value of 0.025 kg meat kg^{-1} milk to yield an allocation of 14.4% to meat and 85.6%

to milk is provided, or a specific allocation factor can be determined using the meat and milk yield generated by the specific farming system.

An example of the effects of allocation method on the GHG emissions per kilogram of milk and meat is shown in Table 14.1 for a dairy production system in Canada in which the meat was from cull cows and grain-fed veal calves (males and females not used as replacements) finished at 6.5 months of age at 270 kg (McGeough *et al.*, 2012).

Environmental Impacts

Livestock production is closely linked to the environment and LCA can be a useful way of assessing the impact of this relationship. A number of environmental impacts are of interest for livestock systems, including global warming potential (assessed as GHG emissions), acidification potential (AP), eutrophication potential (EP), abiotic depletion, desiccation, odour, water resources, land competition and others as shown in Table 14.2. To date, most livestock LCAs have focused on GHG emissions, but there is increasing interest to broaden the analysis for ruminant production systems. Ruminants are a significant source of GHG due to enteric CH_4 from ruminal fermentation of cellulolosic feedstuffs, yet grazing ruminants can also have many ancillary environmental benefits such as helping to preserve forage lands that sequester soil C reserves, thereby withholding CO_2 from the air (Garnett, 2009). These grazing lands have many other ecosystem services including the conservation of biodiversity, water quality, wildlife habitat and aesthetic value (Janzen, 2011). The challenge when incorporating several impact categories in a LCA is deciding on the appropriate weighting of the various impact categories in order to determine the ecological advantage of a production system (Haas *et al.*, 2001).

Greenhouse Gases

Greenhouse gas life cycle assessment methodology

Animal agriculture generates about 8–10% of global anthropogenic emissions, or higher (16–18%) if emissions from land use change are considered (Steinfeld *et al.*, 2006; O'Mara, 2011). Given the effects of increasing GHG levels on climate change and the growing demand for food to meet population increases, mitigation of these emissions has been a focal point in agricultural research (Garnett, 2009). Many governments are grappling with ways of reducing GHG emissions from agriculture and significant research is now being directed towards developing animal husbandry practices that lower enteric CH_4 emissions from ruminants (McAllister and Newbold, 2008; Beauchemin *et al.*, 2009; Eckard *et al.*, 2010).

LCA can be a useful way of determining the net impact of a particular mitigation strategy on the total GHG emissions per unit of product produced, because it accounts for all changes

Table 14.1. Greenhouse gas emissions as affected by allocation method (from McGeough *et al.*, 2012).

		Emissions per functional unit (kg CO_2e)		
Allocation method[a]	Emission allocation to milk (%)	Milk (kg FPCM[b])	Meat (kg live weight)	Meat (kg carcass weight[c])
No allocation	100	0.91	0	0
Economic	91	0.83	1.72	2.87
Dairy versus beef	97	0.88	1.16	1.94
IDF default	86	0.78	2.84	4.73
IDF specific	73	0.67	5.24	8.73

[a]No allocation, 100% to milk; Economic, based on 5-year average of milk and meat prices; Dairy versus beef, emissions allocated; IDF default, International Dairy Federation equation using default meat to milk ratio; IDF specific, International Dairy Federation equation using the actual meat to milk ratio for the study. [b]Fat and protein corrected milk. [c]Calculated as 0.60 of live weight. Included meat from culled cows, all male and female calves not used as replacements, all finished as grain-fed veal at 6.5 months of age at 270 kg.

Table 14.2. Example impact categories that could be considered in life cycle assessment of livestock production (compiled from Haas *et al*., 2001; Guinée *et al*., 2002; Cederberg and Stadig, 2003).

Impact category	Description	Potential characterization method	Indicator unit
Global warming potential	Production of greenhouse gases	Estimation of CH_4, N_2O and CO_2 using IPCC (2006) guidelines	kg CO_2e
Land competition	Loss of land as a resource due to land use	Aggregation of land use	m^2 $year^{-1}$
Landscape image	Subjective assessment of the aesthetic value of the landscape	Visual assessment	index (1–5)
Biodiversity	Effects on biodiversity resulting from harvesting biotic resources or alteration of land	Based on a statistical measure of plant and animal species density	Species per ha^2 (in development)
Loss of life support function	Resulting from interventions such as harvesting biotic resources or destruction of land	Based on net primary production	in development
Desiccation	Caused by water shortage due to groundwater extraction or manipulation of the water table	Not well developed	in development
Acidification	Emissions of the acidifying pollutants SO_2, NO_x and NH_x to the air	AP of each emission	kg SO_2e
Eutrophication	Emissions of N and P to air, water and soil	EP of each emission	kg PO_4e or kg NO_3e
Odour	Odorous substances	Based on odour threshold values	m^3 (air)
Non-renewable energy	Exhaustion of energy supplies due to direct and indirect energy use in farming	Based on energy used directly, and imported fertilizers	MJ
Water Use	Depletion of water resources	Based on water used for irrigation of feed crops and for drinking by livestock production	litres

AP, acidification potential; EP, eutrophication potential.

in GHG emissions arising from the prospective mitigation practice. Sometimes reducing GHG emissions in one part of a farming system can lead to an increase in emissions in another part of the system, which can only be detected using a whole farm approach (Janzen *et al*., 2006). The total quantity of GHG emissions associated with a product is sometimes referred to as its GHG intensity (Henderson *et al*., 2011) or C-footprint (Kitzes and Wackernagel, 2009), which is expressed as kg CO_2e per kg of product. A measure of GHG intensity avoids favouring practices that reduce emissions at the expense of animal productivity. While GHG intensities of beef and dairy production are reported as a value rather than a range, it should be noted that there is considerable uncertainty associated with these estimates due to the uncertainty in the emissions factors used to estimate CH_4 and N_2O emissions (Flysjö *et al*., 2011b).

Whole farm systems models can be used in LCA to estimate the total GHG emissions (CH_4, N_2O and CO_2) from producing meat, milk and eggs (Crosson *et al*., 2011). The various GHG are expressed as CO_2e to account for the global warming potential of the respective gases (International Panel on Climate Change (IPCC), 2007; 100-year timeframe): CH_4, 25; N_2O, 298; and CO_2, 1. A number of models and software tools have been developed to estimate GHG emissions from farming systems, or their components (e.g. animals, crops, soils), with some of these listed in Table 14.3.

Most GHG models are based to some extent on IPCC (2006) methodology, which provides

Table 14.3. A partial list of models used to predict greenhouse gas emissions from livestock production.

Model	Details	Reference
BEEFGEM	Whole farm beef GHG emissions model used in Ireland, Excel based	Foley *et al*. (2011)
DairyGHG	Software tool for estimating GHG emissions and carbon footprint of dairy production systems in the USA. Includes all major sources and sinks of CH_4, N_2O and CO_2. Http://www.ars.usda.gov/Main/docs.htm?docid=17355	Rotz *et al*. (2010)
DairyWise	Empirical model for estimating GHG from dairy farms, accounts for livestock and feed management, land and crop management; GHG calculated according to the Dutch inventory methodology	Schils *et al*. (2007)
FarmGHG	Models of C and N flows and GHG on dairy farms in Denmark. Quantifies all direct and indirect emissions of CO_2, CH_4 and N_2O; includes imports, exports and flows of all products through the farm	Olesen *et al*. (2004)
Holos	Software tool for estimating the GHG emissions from farms in Canada. All major sources and sinks of CH_4, N_2O and CO_2 and all major livestock and cropping systems considered. Http://www.agr.gc.ca/Holos-ghg	Little *et al*. (2008)
OVERSEER	Software tool to develop nutrient budgets for farms in New Zealand. GHG estimates are based on New Zealand's national inventory methodology	Beukes *et al*. (2010)
PLANETE	Excel-based spreadsheet model from France to estimate direct and indirect GHG linked to farm activities using IPCC tier 2 methodology	Bochu (2002)
$SIMS_{DAIRY}$	Farm-scale model for use on dairy farms in the UK; accounts for GHG, nutrient loss, farm profitability and attributes of biodiversity, milk quality, soil quality and animal welfare	Del Prado *et al*. (2011)

guidelines for quantifying emission sources and sinks from agriculture. Although the IPCC (2006) methodology is designed for national GHG inventory, it can be applied at an industry-wide scale, or at a regional- or farm-scale for use in LCA. The IPCC uses a three-tiered approach with each successive tier having an increased level of detail and accuracy. This tiered approach recognizes the considerable variation in data availability, technical expertise and inventory capacity across countries. Tier 1 is a simple approach that uses yearly default emission factors by animal category. Tier 2 is similar to tier 1, but uses country-specific detail for animals, diets and emission factors as determined by peer-reviewed scientific research. Tier 3 estimates are based on process-oriented models that simulate emissions in high detailed spatial and temporal resolution, based on the input of climatic driving factors and recorded agricultural management. An example of how IPCC methodology might be used in farm-based GHG modelling is Holos, a model developed by Agriculture and Agri-Food Canada (http://www.agr.gc.ca/holos-ghg) to estimate whole-farm GHG emissions. Holos is based broadly on IPCC methodologies, with the algorithms and emission factors modified to reflect Canadian conditions and farming practices. The model considers all significant emissions and removals on the farm, as shown in Fig. 14.2. The model estimates whole-farm GHG emissions, including emissions of enteric CH_4 from ruminants, manure-derived CH_4 and N_2O, soil-derived N_2O, CO_2 from on-farm energy use and the manufacturing of fertilizer and herbicide, and CO_2 emission/removal from management-induced changes in soil C stocks. This whole systems approach can be used to estimate the impact of changes in management practices on whole-farm emissions and has been used in LCA of beef (Beauchemin *et al*., 2010, 2011) and dairy production (McGeough *et al*., 2012).

Methane

Methane is the predominant GHG emission in ruminant livestock systems (Beauchemin *et al*., 2010; McGeough *et al*., 2012) and enteric fermentation within the rumen contributes the

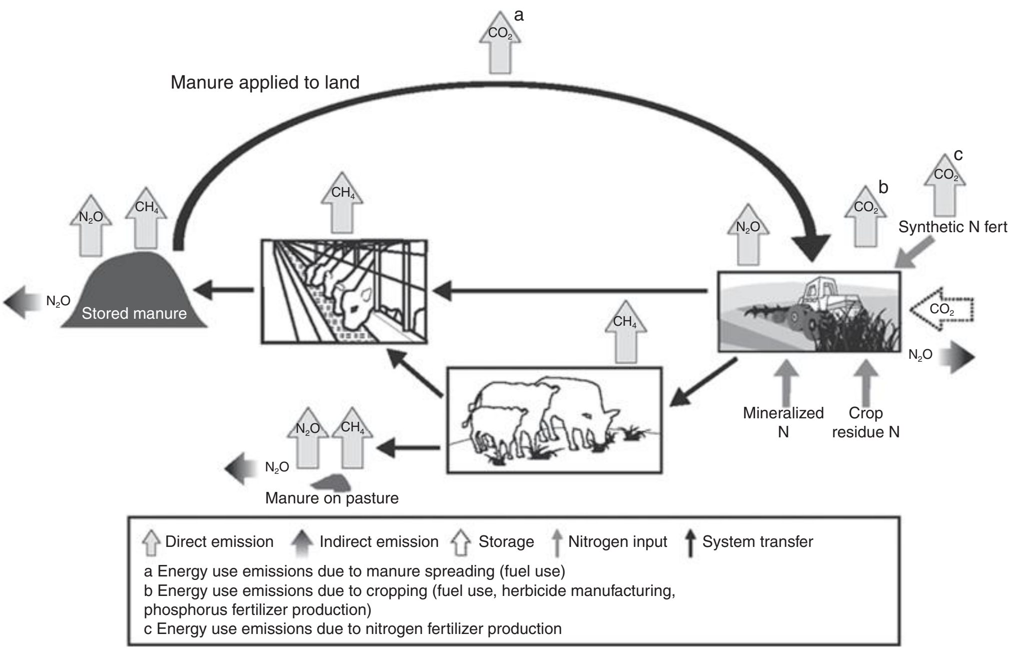

Fig. 14.2. An example of how greenhouse gas emissions and removals can be estimated for a beef farm using the whole-farm model Holos (http://www.agr.gc.ca/holos-ghg). The model calculates enteric CH_4, manure-derived CH_4 and N_2O, emissions for soil-derived N_2O, CO_2 from on-farm energy use and the manufacturing of fertilizer and herbicide, and CO_2 emission/removal from management-induced changes in soil carbon stocks (Little *et al.*, 2008).

majority of this emission. Additionally, CH_4 arises from manure, and accounts for a significant proportion of emissions for non-ruminant production, especially where liquid manure storage systems predominate.

Enteric CH_4

The IPCC (2006) tier 1 approach to estimating enteric CH_4 uses default emission factors (kg CH_4 per animal per year) that differ for dairy cows and other cattle and by geographical location, while the more advanced tier 2 and 3 approaches account for differences in animal productivity and feed quality. Tier 2 methodology estimates enteric CH_4 emissions by calculating gross energy intake (GEI) of the animal, which is then multiplied by a CH_4 conversion factor (Ym). The GEI of the animal can be estimated from dry matter intake (DMI) or if the intake is unknown, from the net energy requirements of the animal for maintenance, activity, growth, pregnancy and lactation as appropriate. The GEI required to meet energy requirements is then estimated taking into account the energy density of the diet. The default Ym value (IPCC, 2006) for mature dairy and beef cattle, sheep and buffalo is 0.065 (range: 0.055–0.075), with the exception of feedlot cattle consuming diets containing more than 90% concentrate, in which case the Ym is 0.03 (range: 0.02–0.04). When highly digestible feed is used, the lower bounds of the Ym range should be utilized and conversely, when feed with lower digestibility is used, the higher bounds are more appropriate.

Using a tier 2 approach, enteric CH_4 can be predicted using country-specific methodology. A number of empirical prediction equations that account for differences in DMI of the animal and the chemical composition of the ration have been proposed, with some listed in Table 14.4. Although many prediction equations are available, none is entirely satisfactory because when evaluated using independent databases, most have been found to have a high prediction error (Ellis *et al.*, 2010; Alemu *et al.*, 2011).

Table 14.4. Some equations used to predict enteric methane emissions from ruminants.

Source	Equation
Blaxter and Clapperton (1965)	CH_4 (MJ day^{-1}) = 5.447 + 0.469 × (energy digestibility at maintenance intake, % of GEI) + multiple of maintenance × (9.930 – 0.21 × (energy digestibility at maintenance intake, % of GEI)/100) × GEI
Ellis *et al*. (2009) (Eq G – Linear)	CH_4 (MJ day^{-1}) = –1.01 + 2.76 × NDF (kg day^{-1}) + 0.722 × starch (kg day^{-1})
(Eq H – Linear)	CH_4 (MJ day^{-1}) = 2.26 + 5.02 × sugar (kg day^{-1}) + 0.0236 × forage (%)
(Eq I – Linear)	CH_4 (MJ day^{-1}) = 2.72 + 0.0937 × MEI (MJ day^{-1}) + 4.31 × cellulose (kg day^{-1}) – 6.49 × hemicellulose (kg day^{-1}) – 7.44 × fat (kg day^{-1})
Jentsch *et al*. (2007)	CH_4 (kJ) = 1.62 × digestible crude protein (g) – 0.38 × digestible crude fat (g) + 3.78 × digestible crude fibre (g) + 1.49 × digestible N-free extract (g) + 1142
Mills *et al*. (2003) (Linear 1)	CH_4 (MJ day^{-1}) = 5.93 + 1.92 × DMI (kg day^{-1})
(Linear 2)	CH_4 (MJ day^{-1}) = 8.25 + 0.07 × MEI (MJ day^{-1})
(Linear 3)	CH_4 (MJ day^{-1}) = 7.30 + 13.13 × N (kg day^{-1}) + 2.04 × ADF (kg day^{-1}) + 0.33 × starch (kg day^{-1})
(Linear 4)	CH_4 (MJ day^{-1}) = 1.06 + 10.27 × dietary forage proportion + 0.87 × DMI (kg day^{-1})
Moe and Tyrrell (1979)	CH_4 (MJ day^{-1}) = 0.341 + 0.511 × NSC (kg day^{-1}) + 1.74 × hemicellulose (kg day^{-1}) + 2.652 × cellulose (kg day^{-1})

MEI, metabolizable energy intake; EE, ether extract; GEI, gross energy intake; NSC, non-structural carbohydrates (starch+sugar).

Manure CH_4

The CH_4 emission from manure is a function of the amount of manure produced by the animal and the portion of the manure that decomposes anaerobically. Anaerobic conditions occur mainly when manure is stored in liquid-based systems such as lagoons, ponds, tanks or pits. Conditions tend to be more aerobic when manure is stacked or piled, or when it is deposited on pastures and rangelands, and thus the CH_4 emissions are lower in this case. The IPCC (2006) tier 1 methodology estimates CH_4 emissions from manure by livestock subcategory using default emission factors that depend on average annual temperature. These default emission factors represent the range in manure volatile solids content and in manure management practices that would be used in each region, as well as the difference in emissions due to temperature. Tier 2 is a more complex method for estimating CH_4 emissions from manure management requiring information on animal and diet characteristics and manure management practices. The production of volatile solids in the manure is calculated from GEI of the animal and digestibility of the diet, or alternatively volatile solids can be estimated using laboratory methods. Volatile solid production is then multiplied by the maximum CH_4 producing capacity of the manure, which varies by animal species and diet composition, and then by the CH_4 conversion factor (MCF). The MCF is specific to the manure management storage system and the average annual temperature of the location. The MCF ranges from <2% for solid manure storage systems in temperatures <14°C to 80% for aerobically stored manure in temperatures >28°C (such as liquid slurry without natural crust cover, uncovered anaerobic lagoons, pit below the animal confinement), thus temperature and storage facility have a large impact on the emission.

Nitrous oxide

Manure N_2O

Nitrous oxide is emitted both directly and indirectly from livestock manure (urine and faeces) during storage and when applied to soils and crops or deposited on pasture during grazing. The direct emission of N_2O from manure depends on the N and C content of manure, and on the duration of the storage and type of treatment

with well aerated manure storage systems producing higher emissions. The emission of N_2O from manure requires an anaerobic environment preceded by aerobic conditions as it is caused by nitrification and denitrification of N. Nitrification is the aerobic process whereby ammonium is oxidized to nitrate; denitrification is the anaerobic process whereby nitrites and nitrates are transformed to N_2O and N_2 gas.

The IPCC (2006) tier 1 method of estimating direct N_2O emissions involves estimating the total amount of N excreted by livestock species by category for each type of manure management system and multiplying by an emission factor for that type of manure management system. The tier 2 method follows the same calculation as tier 1 but includes the use of country-specific data for some or all of these variables, while a tier 3 method would attempt to estimate amount, timing and spatial distribution of livestock excretions as well as their interaction with simulated soil processes (e.g. soil moisture and nitrogen) to calculate N_2O emissions.

Nitrogen is also lost indirectly from manure by volatilization (e.g. NH_3, nitrogen oxides (NO_x)) or to ground or surface water by leaching and runoff. This N is also subject to nitrification and denitrification after loss from the farm, producing 'indirect' N_2O emissions. The IPCC (2006) methodology calculates volatilized N from an estimate of the amount of N excreted, the proportion of manure managed in each manure management system and the fraction of volatilized N. The tier 1 method is applied using default N excretion data, default manure management system data and default fractions of N lost from manure through volatilization and leaching. The tier 2 and 3 approaches, as indicated before, use more elaborated or advanced approaches of estimating some or all of the variables.

Soil N_2O

As with manure, N_2O is also produced in soils through the processes of nitrification and denitrification. The main factor affecting N_2O soil emissions is the availability of inorganic N in the soil. Direct N_2O emissions result from the application of N inputs to soils, and indirect N_2O emissions from the re-deposition of N arising from volatilization, leaching and runoff. The IPCC methodology estimates direct emissions of N_2O from managed soils separately from indirect emissions. Direct N_2O emissions from managed soils are estimated using three emission factors, with default values used in the tier 1 approach and country-specific emission factors that account for differences in land cover, soil type, climatic conditions or management practices used in the tier 2 approach. The first emission factor is for the amount of N_2O emitted from N in soils due to application of inorganic fertilizer and manure, decomposition of crop residues, and mineralization of native soil organic matter. The second emission factor is for the amount of N_2O emitted from organic soils, and the third factor estimates the amount of N_2O emitted from urine and faeces N deposited by grazing animals on pasture. Indirect N_2O emissions are emitted off-farm from N lost via runoff, leaching and volatilization. These emissions are estimated from the assumed fractions of N lost from manure, residues and fertilizer.

Carbon dioxide

Carbon dioxide is usually considered cycled through the atmosphere: CO_2 is taken up by plants and converted to carbohydrates, plants are consumed by animals, C in manure is returned to the soil to decompose. Thus, the CO_2 from animal respiration (i.e. biogenic CO_2) is not usually included in LCA of GHG emissions from farms. The level of organic C in agricultural soils reflects the balance between inputs and removals. Carbon can be released from or sequestered by soil depending upon management practices. For example, practices such as the application of limestone, dolomite or urea fertilizers lead to CO_2 emissions. In addition, tillage of soil leads to a loss of C because organic matter decomposition is accelerated and harvest of the crop results in less C returning to the soil. Some practices that increase organic matter and C in soils include reducing tillage, restoring grasslands, planting perennial crops and eliminating fallowing of land (Janzen *et al.*, 1998).

Most LCAs of ruminant production systems assume soil C is at steady state (i.e. neither gaining nor releasing C). In soils not at steady state, CO_2 emissions and removals from soil C change can dominate the GHG balance of the farm. Beauchemin *et al.* (2011) showed that when grassland was newly seeded on previously cropped

land, its soil C gain more than offset all GHG emissions from a beef production system. Similarly, Liebig *et al.* (2010) reported that even moderately and heavily grazed long-term native grasslands were net CO_2e sinks, even after accounting for the enteric CH_4 produced by the cattle.

Estimating soil C stocks in agricultural land is not easy because soils vary considerably and changes in soil C occur very slowly (over decades), making changes difficult to measure. The IPCC (2006) determines changes in soil C stocks from soil geographical databases and land-use and management for a given country. Tier 1 uses defaults and reference C stocks, tier 2 replaces default values with country-specific estimates and tier 3 estimates changes in soil C from complex models (McGill, 1996).

Greenhouse gas life cycle assessment of dairy production

A number of farm-based LCAs of GHG emissions from dairy production have been published in recent years (Cederberg and Mattsson, 2000; Haas *et al.*, 2001; Phetteplace *et al.*, 2001; Cederberg and Stadig, 2003; Casey and Holden, 2005a, b; Schils *et al.*, 2005; Lovett *et al.*, 2006, 2008; Olesen *et al.*, 2006; Williams *et al.*, 2006; Vergé *et al.*, 2007; Thomassen *et al.*, 2008b; Basset-Mens *et al.*, 2009; Beukes *et al.*, 2010; Castanheira *et al.*, 2010; Gerber *et al.*, 2010, 2011; O'Brien *et al.*, 2010; Rotz *et al.*, 2010; Bell *et al.*, 2011; Kristensen *et al.*, 2011; Vellinga *et al.*, 2011; McGeough *et al.*, 2012) with most of the earlier ones summarized by Crosson *et al.* (2011).

In an industry-wide analysis of global dairy production, Gerber *et al.* (2010) reported an average GHG intensity (cradle to farm gate; kg CO_2e kg^{-1} FPCM) of 2.4, ranging from 1 to 2 for the industrialized regions of the world to the highest value of 7.5 for sub-Saharan Africa. In this analysis, the allocation of GHG from culled stock was done on the basis of protein, emissions from surplus calves fattened for meat were entirely attributed to meat production, and emissions during pregnancy were entirely attributed to milk production. Changes in C stocks as a result of land use change in the previous 20 years were also accounted for using IPCC (2006) methodology. Slightly lower intensities of milk production for North American systems have been reported by Rotz *et al.* (2010; reported as 0.69 but converted to 0.99 CO_2e kg^{-1} ECM after removing biogenic CO_2 and soil C sequestration) and by McGeough *et al.* (2012; 0.83 CO_2e kg^{-1} FPCM, no changes in soil C stocks) with about 90% of the total emissions allocated to milk in both studies. Differences in methodology, especially method of allocation to milk versus beef, FU (kg CO_2e kg^{-1} milk, ECM or FPCM) and changes in C stocks, make it very difficult to compare GHG intensities across studies.

Methane is the largest contributor to the GHG intensity of milk (45–70%; Beukes *et al.*, 2010; Castanheira *et al.*, 2010; Bell *et al.*, 2011; Kristensen *et al.*, 2011; McGeough *et al.*, 2012). Of this, >75% is derived directly from enteric fermentation (Rotz *et al.*, 2010; McGeough *et al.*, 2012). It is therefore apparent that abatement of enteric CH_4 would result in the most significant reduction in GHG emissions from both confinement and pasture-based dairy production systems. Emissions of N_2O account for about 20–40% of the GHG intensity of milk (Rotz *et al.*, 2010; Flysjö *et al.*, 2011b; Kristensen *et al.*, 2011; McGeough *et al.*, 2012). The relatively large contribution of N_2O indicates that there is considerable opportunity to reduce the GHG intensity of milk production through diet formulation to reduce the amount of N excreted by animals and by adopting alternative manure management practices or reducing the use of inorganic fertilizers (Olesen *et al.*, 2006; Rotz *et al.*, 2010).

A number of strategies to reduce the GHG intensity of milk production have been proposed, with some of the more promising approaches discussed below.

Production Efficiency

Productivity of individual dairy cows has increased tremendously over the years in many parts of the world due to genetics, nutrition and management (see Chapter 2, this volume, for further details). These improvements in animal productivity and efficiency have helped to reduce the GHG intensity of milk production, because emission intensity is inversely related to productivity (Gerber *et al.*, 2011). For example, Capper *et al.* (2009) reported that modern US dairy farms produce the same quantity of milk as dairy farms in

1944, but with 21% of the animals, 23% of the feedstuffs and only 10% of the land. As a result, the GHG intensity of milk production (cradle to farm-gate) is estimated now to be 37% of what it was back then (1.35 versus 3.66 kg CO_2e kg^{-1} milk). Decreased GHG intensity with increased milk production is achieved mainly through increased efficiency caused by a dilution of maintenance energy requirements per kg of milk (Place and Mitloehner, 2010). Thus, increased productivity of cows offers considerable mitigation potential for dairy production systems characterized by low productivity.

In addition to increased milk production, improved feed utilization can be a highly effective way of reducing CO_2e emissions per kg of milk (Vellinga *et al.*, 2011). Bell *et al.* (2011) showed that management and breeding of dairy cows to improve feed use within a system decreased the need for purchased feeds and fertilizers. Similarly, Kristensen *et al.* (2011) observed in an analysis of commercial dairy farms in Denmark, that herd efficiency explained most of the variation in the range of GHG intensities of milk among farms, with emission being 0.13 kg CO_2e kg of ECM lower for the most efficient group compared with the least efficient group. This was mainly due to a higher conversion of feed to milk of 1.32 versus 1.06 kg milk per kg DMI, which reduced the CH_4 emission, and to some extent due to the higher milk yield of 8488 versus 6964 kg ECM per cow.

Nutrition

Diet manipulation is the most direct, and arguably the most effective, means of lowering the GHG intensity of milk for intensive dairy operations because the diet composition directly affects CH_4 emissions. A number of dietary strategies have been evaluated for their effects on GHG intensity of milk and most studies agree that increasing the proportion of forage in the diet increases the GHG intensity of milk production (Rotz *et al.*, 2010; Bell *et al.*, 2011). For example, Rotz *et al.* (2010) reported for the US dairy system that increasing the proportion of the forage in the diet of lactating cows from 0.45 to 0.48–0.70 of the dry matter (depending on stage of lactation) increased the GHG intensity by 17%, due to increased enteric CH_4 and manure emissions. Additionally, harvesting forage requires more machinery and greater fuel consumption compared with feed grain. The same study showed that the type of forage was also important. Increasing the use of maize silage and decreasing the use of lucerne silage lowered the GHG intensity of milk production by 13%. Use of forages that contain high quantities of starch lowers CH_4 emissions (Grainger and Beauchemin, 2011). A number of other nutritional strategies such as incorporating sources of fat into the diet and supplementation with various feed additives have been proposed as a means of lowering methanogenesis in the rumen (Johnson and Johnson, 1995; Beauchemin *et al.*, 2009; Martin *et al.*, 2010; Grainger and Beauchemin, 2011). However, few of these strategies have been explored in detailed for their net effect on GHG intensity of milk production.

Pastoral versus Confined Dairy Systems

Farm-based LCAs report that GHG intensity of dairy production (kg CO_2e kg^{-1} ECM) in pastoral systems is similar to (Flysjö *et al.*, 2011b) or higher than reported for confined systems (Beukes *et al.*, 2010; Bell *et al.*, 2011). Differences are partly attributed to higher enteric CH_4 production of cows fed high forage diets. Diets with high amounts of readily fermentable carbohydrates reduce enteric CH_4 production relative to that in high-fibre pastoral diets, due to greater propionic acid production and lower pH in the rumen and the subsequent inhibitory effects on methanogenesis (Beauchemin *et al.*, 2009). Additionally, the difference is due to level of animal performance; milk yield in intensive dairy systems is often higher than that reported in pastoral systems (e.g. Beukes *et al.*, 2010; Flysjö *et al.*, 2011b).

In pastoral systems, both milk production and GHG emissions are driven by stocking rate, through effects on feed intake. Thus, the challenge for pastoral systems is mitigating GHG without causing a decline in milk production. Beukes *et al.* (2010) investigated various potential GHG mitigation options for the pastoral

dairying system in New Zealand and concluded that there is potential to decrease GHG intensity of milk production by up to 34%. This could be achieved through a combination of strategies: improvements in reproductive performance leading to less culling, use of high genetic merit cows, and better pasture management to permit lower stocking rates and less N fertilizer use. Haas *et al*. (2001) explored grassland dairy systems in Germany and concluded that extensive systems had a lower GHG intensity than intensive or organic farming systems. The extensive farms used lower stocking rates, less N fertilizer and produced less milk per cow than the intensive farms. In a study of pastoral dairy systems, O'Brien *et al*. (2010) evaluated three divergent lines of Holstein Friesian cows (i.e. high production North American, high durability North American and New Zealand line) in three feeding systems (i.e. high grass allowance, high stocking rate and high concentrate supplementation). The most profitable combination was the New Zealand line in the high stocking rate system. This combination also resulted in a 6% reduction GHG intensity of milk (cradle to farm-gate), demonstrating that well managed grass-based dairy production systems can achieve high profitability and low GHG emissions.

Reproductive Performance

Modern high-producing dairy cows are considered sub-fertile, as characterized by low pregnancy rates and high rates of embryonic mortality, and this situation has a major impact on the GHG intensity of milk production (Garnsworthy, 2004). About a quarter of the emissions are from replacement heifers in intensive dairy operations (McGeough *et al*., 2012), so it follows that improvement in cow fertility and a resulting decrease in the number of replacements needed to maintain herd size for a given level of milk production would decrease the GHG intensity of milk production. Garnsworthy (2004) showed for an intensive UK dairy system that total enteric CH_4 emissions from replacements could be reduced by 24% by improving fertility to an ideal level through improved oestrous and conception rate of cows.

Herd Health and Animal Management

The health challenges of modern dairy production not only affect farm profitability, but increase GHG intensity of milk production because the quantity of milk shipped from the farm decreases, yet emissions remain unchanged. Mastitis, metabolic diseases, lameness, transition cow management, grouping strategies and calf management have been identified as potential improvement areas in intensive dairying that would lead to lower GHG intensity of milk production (Place and Mitloehner, 2010).

Greenhouse Gas Intensity of Beef Production

Beef production systems are often very complex, which can be challenging in terms of establishing the system boundary for a LCA. Beef production is typically comprised of farms that produce calves (cow–calf farms) and farms that grow and fatten calves for market. Sometimes the entire process occurs within a single farm, but more often a series of farms is involved with calves moving from farm to farm. The system also typically incorporates varying degrees of grazing and confinement. Many farm-based LCAs of GHG emissions from beef production systems have been published in recent years (Subak, 1999; Phetteplace *et al*., 2001; Cederberg and Stadig, 2003; Johnson *et al*., 2003; Ogino *et al*., 2004, 2007; Casey and Holden, 2006a, b; Williams *et al*., 2006; Stewart *et al*., 2009; Beauchemin *et al*., 2010, 2011; Crosson *et al*., 2010; Nguyen *et al*., 2010; Pelletier *et al*., 2010; Peters *et al*., 2010; Veysset *et al*., 2010; White *et al*., 2010; Foley *et al*., 2011) with most of the ones published by 2010 summarized recently (Crosson *et al*., 2011). LCAs that consider the entire beef production system (i.e. including the breeding stock and growth/fattening of the progeny) report a GHG intensity ranging from 10 to 17 kg CO_2e kg^{-1} live weight (14–37 CO_2e kg^{-1} of carcass weight). The lower intensity estimates are for intensive beef production systems used in North America (Johnson *et al*., 2003; Beauchemin *et al*, 2010; Pelletier *et al*., 2010) and parts of Europe (Casey and Holden, 2006b; Crosson *et al*., 2010;

Veysset *et al.*, 2010). Higher GHG intensities have been reported for a Japanese system that relies on imported grains (14.56 kg CO_2e kg^{-1} live weight, converted from 36.4 kg CO_2e kg^{-1} carcass with 40% dressing percentage; Ogino *et al.*, 2004, 2007) and for pasture-based systems in Brazil (Cederberg *et al.*, 2011; converted from carcass weight assuming a 60% dressing percentage; 17 kg CO_2e kg^{-1} live weight not including land use change and 284 kg CO_2e kg^{-1} live weight after accounting for land use change). The variation in GHG intensity of beef production reflects differences in the beef production systems used in different geographical areas, but also can be attributed to differences in the scope, boundary and methodology of the individual LCAs, which can differ. Thus, it is not justifiable to compare GHG intensity across studies with the aim of identifying more GHG efficient production systems. Rather, an LCA of a particular beef production system can be useful for determining the impact of diet, animal husbandry and other management decisions on GHG intensity for that particular system.

Methane from enteric and manure sources usually represents 55–70% of the total GHG emissions of beef production. The cow–calf phase emits about 75–90% of the total emissions associated with beef production (Johnson *et al.*, 2003; Beauchemin *et al.*, 2010), thus partial LCAs that only examine GHG attributed to the fattening phase are very limited in scope. When examined over a number of studies, some important findings for beef production arise as discussed below.

Production efficiency

Improved animal productivity and enhanced efficiency of production lower GHG intensity of beef production, mainly through increased animal output with more efficient use of resources. Improved animal rate of gain reduces GHG intensity because animals reach market weight at a younger age, and thus are fed fewer days, so emissions over the feeding period decrease without a change in product output. Efficiency of producing beef can be increased through improved herd management (Beauchemin *et al.*, 2011), better reproductive efficiency (Garnsworthy *et al.*, 2004), use of growth promotants (Cooprider *et al.*, 2011) and improved calf survival (Beauchemin *et al.*, 2011). For example, Cooprider *et al.* (2011) reported that use of growth promotants and ionophores during the feedlot phase lowered GHG by 22%, due to increased average daily gain and fewer days to market. Assuming that the feedlot finishing phase accounts for about 12% of the total GHG intensity of producing beef in the North American system (Beauchemin *et al.*, 2010), the GHG intensity of beef production to the farm gate could be reduced by about 2–3% due to the use of these growth promoting technologies. Of course, adoption of such management practices is driven primarily by farm profitability (and legislation in the case of growth-enhancing technologies, which are restricted in some areas) and not GHG intensity.

The impact of improved animal productivity on GHG intensity of beef production is illustrated in an industry-wide study by Capper (2011) that examines the long-term changes in the US industry (from 1977 to 2007). Over the 30-year period, days to slaughter reduced from 602 days in 1977 to 482 days in 2007, while carcass yield increased from 274 to 351 kg during the same period. Increased carcass yield per animal reduced the size of the supporting cow herd required, which in turn reduced resource use. The net result was a 16% decrease in GHG intensity of beef production during this period. Similarly, a study of the Canadian beef industry (including livestock, feed and manure) reported that between 1981 and 2001, GHG emissions per kg of live weight decreased by 36% (from 16.4 to 10.4 kg of CO_2e) (Vergé *et al.*, 2008). This reduction in GHG was attributed to increased technical efficiency and improved animal performance due to genetics, nutrition and management. A LCA of French beef production systems showed a range in GHG intensity (range of 14.6–19.0 kg CO_2e kg^{-1} live weight) with the lowest intensity for an intensive system using maize silage producing 17-month-old bulls and 33-month-old fattened heifers and the highest intensity for a farm selling younger (15–16 months) and lighter weight calves (Veysset *et al.*, 2010). In the latter case, emissions from the breeding stock represented a greater proportion of the total emissions. This is why increasing the live weight at slaughter of the progeny decreased GHG intensity, because the relative proportion of

the total emissions attributed to the breeding stock was reduced. It should be pointed out that feeding cattle to a heavier carcass weight only decreases GHG intensity of beef production when both the cow–calf and fattening system are included in the system boundary. For example, Ogino *et al*. (2004) examined a Japanese beef fattening system that only included the calves. The heavier carcass weight obtained by increasing the feeding length from 26 to 28 months did not offset the additional GHG produced.

Strategies that improve the efficiency and productivity of the cow herd by reducing the use of feed resources or increasing the weaning percentage can have a significant impact on reducing the GHG intensity of producing beef, because the breeding stock contribute the greatest proportion of the C-footprint of beef (Beauchemin *et al*., 2010). However, some have suggested that opportunities for significant further improvement in animal productivity may be restricted mainly to developing countries, because animal productivity is already high in most OECD countries (Henderson *et al*., 2011).

Nutrition

Alterations in diet composition have been proposed as a means of reducing GHG emissions from beef cattle (Johnson and Johnson, 1995; Beauchemin *et al*., 2009; Martin *et al*., 2010; Grainger and Beauchemin, 2011). As previously discussed for dairy cows, improving the nutritive value of the diet can lower enteric CH_4 emissions. In a farm LCA of beef production, Beauchemin *et al*. (2011) showed that feeding more highly digestible forages to the cow herd lowered GHG emission intensity of beef production by 5%. Energy density of the ration affects the quantity of feed required to meet the animals' energy requirements, and enteric CH_4 is proportional to DMI (Grainger *et al*., 2007) and GEI (IPCC, 2006). Thus, feeding energy-dense feeds to mature beef cows can reduce CH_4 emissions, as these animals are typically limit-fed to meet energy requirements.

Another proposed approach to lowering CH_4 emissions has been supplementation of diets with lipids. Beauchemin *et al*. (2011) showed the potential to decrease GHG emission intensity through fat supplementation of diets using canola seed as a source of lipids. The study included the emissions associated with growing the canola seed. Feeding canola seed to the cow calf herd reduced GHG emission intensity by 8%, while feeding it to backgrounding cattle reduced GHG emission intensity of beef production by 1% and feeding it to finishing cattle reduced intensity by 2%. The reduction was due to lower enteric CH_4 emissions because of the reduced DMI of the higher energy diet containing canola and a lower Ym of lipid-supplemented diets.

Grain versus Forage Finishing

Within beef fattening systems, increasing the duration of grain feeding while decreasing the duration of forage feeding (using conserved forages or pasture) decreases GHG intensity, as shown in a number of studies (Table 14.5). Similarly, pasture-finished cattle have a higher GHG intensity than do grain-finished cattle (Pelletier *et al*., 2010; Peters *et al*., 2010; Capper, 2011; Cederberg *et al*., 2011). Forage feeding tends to decrease average daily gain, thereby prolonging the time to slaughter (which increases emissions), but not animal product (live weight) if calves are marketed at the same weight. Furthermore, enteric CH_4 emissions increase because of the higher Ym of forage-based diets compared with grain-based diets, and higher maintenance energy requirements of pastured cattle due to increased activity during grazing. Manure emissions are usually increased because more manure is excreted in the longer time to slaughter and the N_2O emission factor for manure deposited on pasture is higher than that for deep-bedded manure (Beauchemin *et al*., 2011). Thus, while increased grazing of cattle may be advantageous for reasons such as consumer preference, fatty acid profile of the meat, animal welfare, maintenance of pastoral land, cost of production and other reasons, this is not the case for GHG emissions.

Pasture Stocking Rate

In most parts of the world, beef production is partly or entirely pasture-based. Pastoral systems have limited potential to decrease enteric

Table 14.5. Summary of the literature examining grain-based versus forage-based finishing systems for beef cattle on greenhouse gas (GHG) intensity.[a]

Reference	Production system	Days in feedlot on high grain ration	GHG intensity (CO_2e kg^{-1} beef live weight)
Beauchemin *et al.* (2011)	Extended grain finishing	210	12.8
	Backgrounding and finishing	170	13.0
	Reduced grain finishing	120	13.9
Capper (2011)	Conventional feedlot finishing	135	15
	Pasture finished	0	26
Pelletier *et al.* (2010)	Traditional feedlot system	303	14.8
	Reduced grain finishing	150	16.2
	Pasture finished	0	19.2
Peters *et al.* (2010)	Grain finished	115	5.9
	Grass finished	0	7.2

[a]Soil organic carbon was assumed to be at equilibrium in all studies.

CH_4 through diet supplementation, and supplementing with grain should be promoted as a CH_4 mitigation strategy only after careful assessment using LCA and economic analysis. Furthermore, grain feeding ignores the importance of ruminants in converting fibrous feeds, unsuitable for human consumption, to high-quality protein sources such as milk and meat.

Several LCAs have examined the effects of stocking density as a means to reduce GHG emissions of pastoral-based beef production systems. Carbon sequestration was not considered in these analyses because equilibrium was assumed, although it is well recognized that permanent grasslands can act as large C sinks (Liebig *et al.*, 2010). For a traditional Irish beef suckler system, both Casey and Holden (2006b) and Foley *et al.* (2011) reported that where stocking rates were already at moderate levels, further increases had negative implications for GHG intensity (CO_2e emissions per hectare and per live weight). This was because higher stocking rates led to higher fertilizer-related emissions, which were largely responsible for the increased GHG intensity. Similarly for beef and sheep farms in New Zealand, White *et al.* (2010) observed that increased levels of N fertilizer were necessary to support higher stocking densities, which led to increased GHG intensities. While it is recognized that some degree of intensification is warranted in terms of maximizing land use and avoiding land use change (Nguyen *et al.*, 2010), and deforestation in some areas such as Brazil (Cederberg *et al.*, 2011), both the agricultural outputs and GHG emissions as well as the long-term impact on pasture productivity and sustainability (Burrows *et al.*, 2010) need to be considered in the analysis.

Dairy–Beef Systems

Dairy systems produce meat from culled animals as well as surplus calves fattened for meat, and in some parts of the world a significant portion of the beef produced is a co-product from dairy production. According to FAO, as much as 57% of the global cattle meat production is from the dairy sector (Gerber *et al.*, 2010) although the global trend is towards more intensification of milk production per cow, resulting in less meat production per kilogram of milk (Flysjö *et al.*, 2011b). A number of studies have shown that meat from integrated dairy–beef systems has a lower GHG intensity than from traditional beef production systems (Casey and Holden, 2006b; Nguyen *et al.*, 2010). In a dairy–beef system, the GHG emissions from the breeding stock are allocated to both meat and milk, whereas all emissions are allocated to the meat in a traditional beef system, and as stated previously, minimizing the contribution of the breeding stock to the total GHG emissions has a large impact on GHG intensity of beef production. Nguyen *et al.* (2010) compared a traditional beef system (suckler cow–calf) with several

dairy–beef systems used in the EU and showed that the traditional system had a GHG intensity that was 37–70% higher than the dairy–beef systems. In that study, a biological allocation of GHG to meat and milk was used; the feed needed to meet the cows' extra energy requirements due to pregnancy were assigned to the calves and the system boundary did not consider the meat from the culled cow. Casey and Holden (2006b) used mass allocation, which assumed 96.6% of GHG was attributed to milk and 3.4% to meat based on weight at sale of the products. The dairy–beef systems had 13.5–36.3% lower emissions compared with a traditional beef system depending on the type of system used.

Greenhouse Gas Life Cycle Assessment of Sheep Production

Compared with beef and dairy, relatively few LCAs of GHG emissions from sheep production systems have been published (Casey and Holden, 2005c, as cited by Ledgard *et al*., 2011; Williams *et al*., 2008, as cited by Ledgard *et al*., 2011; Edwards-Jones *et al*., 2009; Biswas *et al*., 2010; Peters *et al*., 2010; Alcock and Hegarty, 2011; Ledgard *et al*., 2011). The GHG intensity of sheep meat (cradle to farm gate) reported in those studies ranged from a low of 5.0 kg CO_2e kg^{-1} live weight for a simulated prime lamb enterprise in Australian (Alcock and Hegary, 2011) to 14.1 kg CO_2e kg^{-1} live weight for a UK system (Williams *et al*., 2008, as cited by Ledgard *et al*., 2011). A much higher value of 52.6 CO_2e kg^{-1} live weight was reported for a particular sheep production system in Wales, but this was an unusual case study with peat soils and high N_2O emissions (Williams *et al*., 2008, as cited by Ledgard *et al*., 2011). Generally, the GHG intensity of sheep meat production appears to be somewhat lower than beef (i.e. 5–14 versus 10–22 kg CO_2e kg^{-1} live weight). However, when comparing across studies it is not clear how much of this difference is due to LCA methodology and system boundary. Within study, Peters *et al*. (2010) compared several sheep and beef production systems in Australia and found the sheep system had 70% the GHG intensity of beef systems.

Wool is a co-product from sheep farming, and thus the environmental burden needs to be allocated between meat and wool. In most sheep studies, allocation has been based on economics (e.g. Biswas *et al*., 2010; Ledgard *et al*., 2011).

As with beef and dairy production, enteric CH_4 represents the main contributor to the total GHG intensity of sheep production. Thus, the greatest opportunities for reducing GHG emissions are through reducing CH_4 emissions, which can be done by increasing production efficiency (Ledgard *et al*., 2011). Improved efficiency can be achieved through increased lambing percentages, earlier mating of ewes, improved nutrition to allow lambs to reach slaughter earlier, genetic improvement for faster growth rate and better feed conversion, and finishing lambs at a heavier weight (Alcock and Hegarty, 2011; Ledgard *et al*., 2011).

Other Environmental Impacts

Eutrophication potential

Eutrophication potential (EP) refers to nutrient enrichment that could potentially negatively affect aquatic and terrestrial systems (Finnveden and Potting, 1999). Emissions to the air of nitrogenous compounds and discharges to water of both N and P contribute to eutrophication. In livestock production systems, EP results mainly from soil amendments of inorganic fertilizers and manure and from NH_3 volatilized from barns and animal pens. The proportion of N in plant protein that is converted to and deposited in animal protein is relatively low in farm animals, ranging from 5% to 45% (Oenema and Tamminga, 2005), with utilization by beef cattle the least efficient. Excess N excreted in faeces and urine can lead to NH_3 emissions, increasing the loss of N to the environment. Furthermore, purchasing of feeds is a common practice for most livestock enterprises, resulting in a net importation of P and N on to the farm. Manure spreading can be a significant source of excess P in surface water.

Substances that contribute to EP include NH_3 and nitrogen oxides (NO_x) emitted to the air, N and P emitted to water, and N and P emitted to soil (Huijbregts and Sepp, 2001) with EP calculated according to the method outlined by Guinee *et al*. (2002). Equivalency

factors for the various substances allow EP to be expressed in phosphate (PO_4e) or nitrate (NO_3e) equivalents (Heijungs *et al.*, 1992, as cited by Seppälä *et al.*, 2004).

In a study of beef production systems used in Europe, EP ranged from a high of 1651 g PO_4e kg^{-1} live weight for a semi-extensive suckler cow–calf system to a low of 622 g PO_4e kg^{-1} live weight for a dairy beef system with calves slaughtered at 12 months (Nguyen *et al.*, 2010). Nitrate leaching from soil was by far the most important contributor to EP. Similarly, Cederberg and Stadig (2003) reported a relatively high EP of 1842 g PO_4e kg^{-1} live weight (converted from PO_2e by Nguyen *et al.*, 2010) for a Swedish organic beef production system. The EP of beef production in the Midwestern USA was reported as 104–142 g PO_4e kg^{-1} live weight (Pelletier *et al.*, 2010), with higher emissions associated with pasture-based systems compared with a feedlot system. Higher EP for pasture-based versus feedlot systems occurred as a result of higher feed intake of forage diets, compounded by a significant trampling rate, larger land area required and the high amount of manure produced relative to live weight production. A much lower EP was reported for a Japanese beef production system, possibly because the animal waste was composted (24 g PO_4e kg^{-1} live weight, Ogino *et al.*, 2007). The large range in EP for beef production reported among these various studies may be due to differences in methodologies or varying N and P losses due to diet formulation, manure management and land use.

For dairy production, Haas *et al.* (2001) reported that EP per hectare in southern Germany was highest for intensive farms, intermediate for extensive farms and lowest for organic fairy farms (54.2, 31.2 and 13.5 kg PO_4e ha^{-1}, respectively). Expressing EP on the basis of milk production did not change the ranking for the farm types (7.50, 4.46 and 2.78 g PO_4e kg^{-1} milk, respectively). Thomassen *et al.* (2008b) showed for dairy farms in the Netherlands that EP was higher for conventional (110 g NO_3e kg^{-1} FPCM) compared with an organic system (70 g NO_3e kg^{-1} FPCM). The difference in EP was mainly due to higher off-farm sources of EP for the conventional system. Nitrate accounted for 32% of the EP in the conventional and for 40% in the organic system; phosphate accounted for 53% in the conventional and for 31% in the organic system; and NH_3 accounted for 12% in the conventional and for 25% in the organic system.

Acidification

Acidification is caused by emissions of the acidifying pollutants SO_2, NO_x and NH_x to the air, which can be converted into acids, and cause death of fish and forests, as well as other environmental damage. Ammonia from manure is the most significant source of AP in livestock systems (Cederberg and Stadig, 2003; Nguyen *et al.*, 2010). Acidification potentials are presented in sulfate equivalents (SO_2e) with further detail on methodology given by Guinée *et al.* (2002).

A study of beef production systems in Europe (Nguyen *et al.*, 2010) reported that AP ranged from 101 to 210 g SO_2e kg^{-1} beef live weight, with the highest AP for a traditional suckler beef production system and lowest AP for a dairy–beef system with bulls fattened to 12 months of age. The main contributor (>70%) to AP was direct NH_3 emissions from the pens that occurred during the fattening period. Ogino *et al.* (2007) reported an AP of 136 g SO_2e kg^{-1} beef live weight for a Japanese beef system (after FU was converted from carcass weight). Cederberg and Stadig (2003) reported an AP of 448 g SO_2e kg^{-1} beef live weight (converted from H+ kg^{-1} by Nguyen *et al.*, 2010) for a Swedish organic beef production system.

For dairy production, Castanheira *et al.* (2010) reported that the AP of milk from Portuguese dairies was about 20 g SO_2e kg^{-1} of milk, with 70% attributed to dairy farm activities, mainly as a result of NH_3 emissions from volatilization of N from manure. For dairy farms in Germany, Haas *et al.* (2001) reported that AP was lower for organic and extensive farms compared with intensive farms (107, 119 versus 136 kg SO_2e ha^{-1}), but when expressed on the basis of milk production, the AP was lower for intensive and extensive farms compared with organic farms (18.8, 17.0 versus 22.3 g SO_2e kg^{-1} milk). Thomassen *et al.* (2008b) reported that in the Netherlands AP of conventional dairy farms was 9.5 g SO_2e kg^{-1} FPCM and 10.8 g SO_2e kg^{-1} FPCM for organic farms, with NH_3 volatilization accounting for 74% and 81% of the AP in conventional and organic systems, respectively.

The AP of beef and dairy systems is highly influenced by NH_3 emissions, therefore mitigation of AP can mainly be achieved by reducing the amount of N excreted, particularly urinary N (which is more highly volatile than N in faeces), and furthermore by using improved manure handling systems.

Non-renewable Energy Use

Non-renewable energy use can be another indicator of sustainability of livestock production systems, as it comes from finite resources. Nguyen *et al.* (2010) reported that 41–59 MJ of non-renewable energy was used to produce 1 kg of live weight for beef production in the EU, with 42–43% attributed to fertilizer manufacturing. Much higher non-renewable energy use was reported by Ogino *et al.* (2007) for a Japanese beef production system, because of higher production and transportation energy use (feed from the USA) and a longer fattening period (26–28 months).

Limitations and Future Direction

Currently, the major limitation to LCA for animal products is the inability to compare across studies due to the lack of standardization in methodology coupled with the complexity of farming systems used worldwide. Another limitation of the current methodology for LCA of animal products is that it does not consider the environmental impacts related to a change in land use. The central issue is that in some areas, especially Brazil, expansion of livestock production contributes to deforestation for pastureland and soybean production for animal feeding (Steinfeld *et al.*, 2006). Hence, compelling arguments have been made that land use change (direct effects or through feed importation) needs to be included in LCA of animal products, as this omission underestimates the impact of livestock production on climate change (Cederberg *et al.*, 2011; Weiss and Leip, 2012). While the methodology for including land use change in LCA of animal products is not well developed at this stage, it is likely to happen in the future, which will undoubtedly increase the GHG intensity of animal products, especially in the case of beef.

It must also be recognized that there is a large degree of uncertainty in estimating environmental burden, especially for GHGs due to the uncertainties of the emissions factors used in the analysis. Advances in the understanding of the biological processes of GHG production will reduce the uncertainty associated with GHG emission factors.

To date, the application of LCA in animal production has focused mainly on assessing GHG emissions, because many of the other environmental impacts (Table 14.2) are more difficult to quantify and require further methodological development before they can be used in LCA. Furthermore, by definition LCA focuses on environmental burdens and not the ecological benefits afforded by livestock production. Benefits such as creating food for humans from inedible biomass, conservation of grassland ecosystems, promoting the use of perennial forages that help preserve lands, and recycling of nutrients are not included in LCA of animal production. Thus, there is a need to develop LCA methodology that accounts for the ecosystems services that animal production promotes, particularly in the case of grazing ruminants. The carbon intensity of beef production is the highest of all meat products, in part because beef production systems rely heavily on the use of pasture and grazing lands. In many parts of the world, livestock, beef cattle and sheep in particular, are raised on lands not suitable for other types of food production. Conversion of native grasslands to cultivated land would lead to habitat destruction for wildlife and decreased biodiversity. Thus, focusing LCA only on C-footprint can lead to incorrect recommendations. Future LCA of livestock products need to use multiple impact categories, beyond simply GHG emissions. However, at present there is no clear means of integrating the information for various impacts.

Conclusions

The environmental cost of producing meat and milk must be weighed against the benefits

in terms of food security. Consequently, the livestock industry must continue to improve its efficiency of production, while lowering its environmental footprint. Farm-based LCA offers the livestock industry a structured methodology to assess improved practices and approaches that will help reduce their environmental impact while providing high quality food for human consumption. A cradle-to-farm gate LCA can help identify components or processes within the primary production cycle that should be targeted to reduce environmental impact. Because of the significant variation in methodologies used in LCA, particularly for estimating GHG emissions, caution must be exercised when comparing across studies. Efforts to standardize LCA methodology in future will help improve the usefulness of these types of comparisons.

Clearly, meat and milk production have a 'cost' in terms of GHGs and other environmental impacts, and LCA helps quantify those impacts. Technologies and practices have been identified that can help limit resource use and resulting environmental consequences of meat and milk production, with improvements in productivity, feed conversion efficiency, diet formulation and reproductive performance the first steps in reducing the ecological footprint of livestock production systems.

It must also be recognized that GHG intensity of meat and milk production does not consider the other beneficial roles of ruminant production systems. Ruminants make a useful contribution to society in that they produce high quality energy and protein for human consumption from land areas and cellulosic materials that would otherwise be very difficult to exploit.

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15 Quantitative Measurements of Ammonia and Methane Loss from Livestock

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Introduction

The sustainability of agriculture involves a balance of environmental, social and economic attributes (Cooprider *et al.*, 2011). It follows that a cornerstone of agricultural sustainability is the environmental impact of farming practices on our air, land and water. This chapter focuses on the two major atmospheric pollutants associated with the livestock industry, ammonia (NH_3) and methane (CH_4), and describes the major measurement techniques available to quantify their emissions, as well as reviewing the magnitude of these emissions. This chapter does not include nitrous oxide emissions from livestock manure; however, most of the techniques discussed also have application to nitrous oxide and other gas emissions.

Quantifying NH_3 and CH_4 emissions is fundamental to our understanding of the impact of livestock on air, and the development of best management practices (BMP) to mitigate these emissions. The goal of improving the environmental sustainability of the livestock industry using mitigation strategies will become even more relevant with the expected expansion of the livestock industry due to increased demand for meat and milk by an escalating human population estimated to reach 9 billion by 2050 (Smith *et al.*, 2007). The sources of livestock NH_3 and CH_4 emissions considered in this chapter are the CH_4 associated with enteric fermentation in ruminants, and NH_3 and CH_4 from livestock manure. Previous reviews of the measurement techniques used in agriculture include Harper *et al.* (2011), and Makkar and Vercoe (2007) who focused on *in vivo* CH_4 methane from ruminants. McGinn (2006) and the National Research Council (2003) reported on emission techniques for intensive livestock systems. Previous reviews of techniques to monitor NH_3 losses from livestock manure include McGinn and Janzen (1998) and Shah *et al.* (2006). Neftel *et al.* (2006) reported on the principles for detecting greenhouse gas (GHG) concentration (e.g. optical systems) and some emission techniques with application to agriculture.

Source and Impact of Ammonia

Ammonia is released as a result of the breakdown of protein in the animal feed by microbes that produce microbial protein. When an excess of protein is fed, or when there is a lack of carbohydrates (energy) for microbial growth, the unused NH_3 is excreted.

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Livestock have a low N retention value, where less than 30% of the fed N is used by the animal. Some 50–80% of the unused N is excreted in urine and the remainder in the faeces (McCrory and Hobbs, 2001). As a result of this, livestock manure (urine and faeces) contains a high fraction of the fed N in the animal's diet. However, much of this excreted N is not recycled through application to crop land. About 75% of the manure N from beef cattle is lost as a combination of runoff, NH_3 volatilization and denitrification, prior to its use as a soil amendment for crop growth (Eghball and Power, 1994). The excreted urinary urea is rapidly converted (hydrolysed by microbial urease in the faeces) to NH_3 that is emitted into the atmosphere. The rate of this conversion depends on temperature, pH and moisture content. The conversion to NH_3 in faecal matter is slower to volatilize during storage and land applications. Bierman *et al.* (1999) measured the N content of manure to infer that between 57% and 67% of the excreted N in manure in a beef feedlot was lost as NH_3 volatilization (function of diet). Actual NH_3 emission measurements confirm the large emission of NH_3 from beef feedlots, ranging from 27% in winter to 55% in summer of the fed N as reported by Todd *et al.* (2005). Flesch *et al.* (2007) and McGinn *et al.* (2007) reported 63–65% and 63% of the fed N was lost as NH_3 in a feedlot, respectively.

Ammonia emission continues after manure is applied to cropland. The proportion of N content of the manure emitted as NH_3 to the atmosphere depends on: the type (solid or slurry); manure additives (McCrory and Hobbs, 2001), e.g. acidification (Petersen *et al.*, 2012); application technique, e.g. injected or surface applied (Bittman *et al.*, 2005); and the pre- and post-surface management, e.g. tillage, irrigation (McGinn and Sommer, 2007). In general, reducing the exposure of manure to air reduces the NH_3 emission.

Globally, the release of NH_3 from livestock manure is recognized as the major source of atmospheric NH_3, accounting for 50% of all terrestrial sources (National Research Council, 2003). Once released to the atmosphere, a fraction of the NH_3 is deposited back to the local surface while the remainder reacts with atmospheric acids to form particulate matter (PM) less than 10 μm in diameter. The formation of these fine aerosols can be an air quality issue (McCubbin *et al.*, 2002; Erisman and Schaap, 2004) especially in confined air sheds where high concentrations of PM pose a health risk to people (Popendorf *et al.*, 1985) and animals (MacVean *et al.*, 1986). The formation of fine aerosols can also cause visibility degradation (Barthelmie and Pryor, 1998). The eventual deposition of NH_3 to land can also create problems due to acidification of soil (van Breeman and van Dijk, 1988) and elevated soil nitrogen concentrations causing changes in plant species diversity of natural ecosystems (Sutton *et al.*, 1993).

Source and Impact of Methane

Methane is generated from the fermentation of carbohydrates that occurs during the digestion of feeds in ruminant animals (e.g. cattle and sheep) and during the anaerobic storage of livestock (ruminants and non-ruminants) manure. About 90% of the enteric CH_4 produced by sheep (ruminant) was reported by Murray *et al.* (1976) to originate in the rumen while 10% occurred in the large intestine. However, of this latter 10% generated in large intestine, approximately 90% is absorbed into the blood and is released during respiration. It follows that ruminants eructate and expire the majority (*c.* 99%) of the enteric CH_4 through the nostrils and mouth while only about 1% is lost through the rectum. Although feed is mostly digested in the rumen, many dietary factors increase the proportion of feed digested post-ruminally. For example, decreasing the particle size of feeds, increasing the level of intake, use of treatments such as heating to reduce ruminal degradability, and use of resistant starches, can increase post-ruminal digestion (Galyean and Owens, 1991).

The anaerobic storage of manure in open lots, lagoons and compost piles is also a source of atmospheric CH_4. Leytem *et al.* (2011) investigated the methane emission from dairy wastewater and compost piles of manure and found emissions of 103 and 13.5 g m^{-2} day^{-1}, respectively. Todd *et al.* (2011) reported a mean CH_4 emission from an aerobic dairy lagoon in Texas to be 40 g m^{-2} day^{-1}, or 211 g per animal per day.

Methane from livestock significantly contributes to the build-up of GHG in our atmosphere. Lassey (2008) estimates enteric CH_4

emission is 12–17% of the global CH_4 source, whereas an earlier estimate by Moss *et al.* (2000) indicated enteric CH_4 was 12% of the global, 19% of the anthropogenic and 36% of the agricultural CH_4 emissions. Steinfeld and Wassenaar (2007) reviewed carbon and nitrogen emissions from livestock (by region) and concluded that livestock accounted for 18% of global GHG emissions (i.e. global warming effect) and that livestock enteric CH_4 losses were 'the most important source of anthropogenic CH_4 emissions' at 86 Mt year^{-1}. In a recent review, O'Mara (2011) reported on EPA data (EPA, 2006) that show the enteric CH_4 from ruminants is 89% of the enteric plus manure CH_4 emissions (1929 versus total livestock CH_4 emissions of 2164 Mt of CO_2-equivalent per year).

The emission of CH_4 by ruminants is a loss of energy from the feed that could otherwise be utilized by the animal, and accounts for 2–12% of the gross energy content of the feed as reported by Johnson and Johnson (1995). The range in emissions is due mainly to the level of feed intake and the composition of the diet (Moss *et al.*, 2000; Benchaar *et al.*, 2001). In addition to environmental effects, enteric CH_4 emissions also have direct economical implications for the producer in terms of energy feed efficiency.

Techniques for Monitoring Ammonia and Methane Emissions

There are many techniques available to quantify the emission of CH_4 from enteric fermentation in ruminants (Table 15.1) and CH_4 and NH_3 from manure (Tables 15.2 and 15.3, respectively). The techniques for enteric CH_4 include both *in vitro* (not reported in this chapter) and *in vivo* techniques, where *in vivo* methods encompass the use of enclosures (masks, hoods, whole-animal chambers and tunnels), tracer gases and micrometeorological measurements. For NH_3 the techniques generally used include enclosures and micrometeorological measurements. The technique of boundary layer budgeting (Harper *et al.*, 2011) is not included in this chapter since it applies to a regional scale and is beyond the scope of this review.

Table 15.1. Examples of enteric methane emissions for beef cattle, dairy cows and sheep measured using different techniques. Emissions under the same study are direct comparisons of techniques.

Ruminant	Methane emission (g per animal per day)	Technique	Study
Beef cattle	≈138–155	NSS	Tomkins *et al.* (2009)
Beef cattle	136	BLS	Tomkins *et al.* (2011)
	114	FT-SS chamber	
Beef cattle	86–180 (young heifers)	Tracer – enteric	DeRamus *et al.* (2003)
	165–294 (mature cows)		
Beef cattle	209–242 (different sensor)	BLS	Laubach *et al.* (2008)
	167	FG	
	182	MB	
Dairy cows	296 (dry)–438 (lactating)	FT-SS chamber	Sun *et al.* (2008)
Dairy cows	369–374	Tracer – enteric	Foley *et al.* (2009)
Dairy cows	297 (pens)	BLS	Bjorneberg *et al.* (2009)
	325 (pens and lagoon)		
Dairy cows	299	BLS	Gao *et al.* (2011)
Sheep	16–18 (2- and 1-h sample)	NSS	Goopy *et al.* (2011)
Sheep	43–66	FT-SS chamber	Liu *et al.* (2011)
Sheep	14–19	FT-SS face mask	Wang *et al.* (2007)
Sheep	≈14	FT-SS tunnel	Lockyer and Champion (2001)
Sheep	20	EC	Dengel *et al.* (2011)

NSS, non-steady-state; BLS, backward-time Lagrangian stochastic; FT-SS, flow-through steady-state; FG, flux gradient; MB, mass balance; EC, eddy covariance. ≈, Preceding values indicates units were converted for use in this table using the ideal gas law.

Table 15.2. Examples of methane emissions from different manure storage measured using different technologies. Emissions under the same study are direct comparisons of techniques.

Manure management	Methane emission ($g\ m^{-2}\ day^{-1}$)	Technology	Study
Beef feedlot	0–27 (active aeration)	NSS	Hao *et al*. (2001)
Compost pile	0–54 (passive aeration)		
Dairy wastewater and pen runoff	103	BLS	Leytem *et al*. (2011)
Dairy wastewater and pen runoff	40	BLS	Todd *et al*. (2011)
Dairy lagoon	≈16	Biogas production	Craggs *et al*. (2008)
Swine lagoon	≈416	Biogas production	Craggs *et al*. (2008)
Swine lagoon	143	MB	Park *et al*. (2010)
	205	FT-SS	

NSS, non-steady-state; BLS, backward-time Lagrangian stochastic; MB, mass balance; FT-SS, flow-through steady-state. ≈, Preceding values indicates units were converted for use in this table using the ideal gas law

Table 15.3. Examples of ammonia emissions from different manure storage measured using different technologies.

Manure management	Ammonia emission ($g\ m^{-2}\ day^{-1}$)	Technology	Study
Beef feedlot	3–6	FG	Todd *et al*. (2005)
Beef feedlot	8–9	BLS	Flesch *et al*. (2007)
Beef feedlot	1–7 (different feedlots)	BLS	Denmead *et al*. (2008)
Beef cattle manure on grass sward	0.012–0.036	MB	Salomon and Rodhe (2011)
Fresh feedlot manure	12–16 (different diets)	FT-SS	McGinn *et al*. (2002)
Dairy lagoon	5	BLS	McGinn *et al*. (2008b)
Swine lagoon	1.1–1.8 (different lagoons)	FG	Harper *et al*. (2004)
Swine slurry on short grass	4	TPS	Gordon *et al*. (1988)

BLS, backward-time Lagrangian stochastic; FG, flux gradient; EC, eddy covariance; FT-SS, flow-through steady-state; MB, mass balance; TPS, theoretical profile shape.

Although some methodology standards have been developed for specific monitoring programmes, e.g. the US National Air Emissions Monitoring Study (Heber *et al*., 2008), the majority of emission studies are independent where the protocol for determining emissions is unique to each case. It follows that the comparison of studies can be confounded by instrumentation, emission theory and data management. It is therefore important to revisit the emissions techniques for NH_3 and CH_4 and report comparisons on the derived emissions.

Chamber techniques

There are two general designs of enclosures used to measure gas exchange. The non-steady-state (NSS) design monitors the build-up of the gas inside the chamber over time. In the steady-state (SS) design, the gas inside the chamber is in equilibrium with the source emission. There are two types of SS enclosures, the flow-through (FT-SS) and non-flow-through (NFT-SS) type. The FT-SS design relies on fresh air being drawn through the enclosure where there is no build-up of the gas inside the enclosure. The NFT-SS design has a scrubber associated with the chamber that removes the target gas.

All these enclosures can be used to measure enteric CH_4 from livestock or CH_4 and NH_3 from livestock manure. In the case of whole-animal CH_4 chamber measurements, most common is the FT-SS design, e.g. Grainger *et al*. (2007). However, some of the earlier work measuring CH_4 emission from livestock (Turner and

Thornton, 1966) used NSS design. More recent investigation of NSS design has been conducted by Goopy *et al.* (2011) to measure CH_4 emission from sheep for short periods (1–2 h).

For whole-animal CH_4 emissions, a general disadvantage is the expense of building enclosures and the animal training to ensure stress is not an influencing factor in the emission measurement. Caution must be used when using enclosures for measuring NH_3 emissions from manure since NH_3 is readily absorbed by water that may condense on the inner surface of the chamber. This would cause an under-estimation of NH_3 concentration and therefore NH_3 emissions.

Non-steady-state (NSS)

The emission of gas (F_G; flux in g s^{-1} or flux density in g m^{-2} s^{-1}) from the NSS enclosure design is determined as:

$$\text{(a) } F_G = \frac{C_{t2} - C_{t1}}{t2 - t1} V \text{ or } \text{(b) } F_G = \frac{C_{t2} - C_{t1}}{t2 - t1} \frac{V}{A_S} \quad (15.1)$$

where C_{t2} and C_{t1} is the gas concentration (g m^{-3}) at times $t2$ and $t1$ (s), V (m^3) is the inside volume of the enclosure and A_S is the surface area (m^2). Equation 15.1a is used for measuring CH_4 emissions from whole animals (ruminants) while Eqn 15.1b is for NH_3 or CH_4 from a manure surface. For whole-animal CH_4 emissions, the volume of the animal is subtracted from V (Goopy *et al.*, 2011). As well, caution must be exercised to ensure safe concentrations of CO_2 and O_2 exist inside the enclosure and that the welfare of the animal is not jeopardized. This means that the chamber can only be used for short durations (*c.*1–2 h depending on the design of the enclosure). This is problematic for determining daily emissions. For example, Goopy *et al.* (2011) reported r^2 values of 0.42–0.48 between 2 and 22 h enteric CH_4 emissions in sheep. In both the whole animal and manure surface applications, it is important that the build-up of gases inside the chambers does not suppress the emission from the source. The advantage of the NSS design is the simple construction and portability of the chambers but the disadvantage is that measurements are not continuous (e.g. for a manure source the gas concentrations are made periodically throughout the day for just a few minutes) and therefore there is a need to fill data gaps to estimate the 24-h emission.

Non-flow-through steady-state (NFT-SS)

A widely used chamber for determining NH_3 emissions from manure-amended soils is the use of a NFT-SS chamber. The design of this chamber is reported by Marshall and Debell (1980), which consists of a small volume cylinder covering the source. The top of the cylinder that is exposed to the atmosphere has two sponges treated with an acid that removes ambient NH_3. The outer sponge is for the ambient NH_3 while the inner sponge collects only the NH_3 emitted into the chamber air space from the manure surface. It follows that the amount of NH_3 absorbed by the inner sponge over time is assumed to equal that generated at the surface.

This NFT-SS chamber design was used by Bittman *et al.* (2005) to determine NH_3 emissions from dairy cow slurry applied to pasture. They reported that the technique greatly underestimated the NH_3 emission relative to a micrometeorological technique, a finding they report that other studies also found. However, the precision of the chamber was high such that the treatment effect was similar between techniques. The advantage of this enclosure design is the ease of operation, however, the small area of the enclosures and the short sampling time makes sampling of variable (spatial and temporal) emissions difficult.

Flow-through steady-state (FT-SS)

The FT-SS chamber offers an advantage over NSS chambers since gases are not accumulated inside the chamber allowing more continuous measurements over a longer time. Therefore, unlike the NSS design (for short-term emissions), the FT-SS design can be deployed to capture daily patterns directly. The F_G is calculated from this chamber design as:

$$\begin{aligned} &\text{(a) } F_G = f(C_O - C_I) \text{ or} \\ &\text{(b) } F_G = \frac{f}{A_S}(C_O - C_I) \end{aligned} \quad (15.2)$$

where f is the flow-through rate (m^3 s^{-1}), and C_O and C_I are the exhaust and intake concentration (g m^{-3}), respectively. For whole-animal CH_4 emissions, Eqn 15.2a is used to express the emission in units of g day^{-1}. When the technique is used for CH_4 or NH_3 emissions from a manure surface,

the procedure is to express the emission on a per unit area of the surface basis (i.e. Eqn 15.2b).

The unique characteristic of using the FT-SS chamber is that the monitored gas concentration in the air stream passing through the chamber is associated with a known and constant air volume contained by inlet and exhaust ducts. When initially designing the chamber, it is essential that the exhaust concentration (C_O) is appropriate for the gas sensor concentration range; this yields the optimum sensitivity of the instrument. This is accomplished by assuming relevant values for F_G, f and C_I in Eqn 15.2. It is also useful to calculate the time constant of the chamber, i.e. the time it takes for the chamber air to reach 63% of a step change in emission. A value of 3 times the time constant is needed to reach 99% equilibrium of the step change. For example, opening the chamber door when accessing the animal causes a drop in chamber CH_4 concentration (as reflected in the exhaust concentration). Once the door is closed, there is a build-up of CH_4 inside the chamber that will require a time equal to 3 times the time constant to reach near steady-state once again. The time constant of a chamber acts as a filtering function, in that it smoothes the response time of the chamber to changes in the source strength.

The FT-SS design is applied to measure CH_4 emissions from ruminants using: (i) face masks; (ii) head hoods; (iii) whole-animal enclosures; (iv) tunnels; and (v) barns treated as enclosures. These enclosures are typically calibrated by releasing a gas at a known rate, e.g. using a mass flow controller, and verifying the emission using Eqn 15.2. For example, Place *et al.* (2011) developed a head hood for cattle that was shown to recover 98–99% of a known release rate of gas. The release-recovery data can be used to remove the between-enclosure differences when using more than a single enclosure. That will improve the sensitivity of the experimental design to treatment differences. The use of a Latin square experimental design where more than one chamber is used (where each enclosure is used in turn for each treatment) can also be used to remove between-enclosure differences, thus further improving the sensitivity of enclosure measurements to treatment differences.

Biogas production

A simple technique is a commonly used approach to monitor gas production (e.g. biogases) from anaerobic lagoons. In this design, the active gas production is measured by directly monitoring the total gas volume produced over time, and percentage composition of the gas (gases) in the passive off-venting from a chamber on the surface of a lagoon. This approach was used by Craggs *et al.* (2008) using a surface chamber to measure CH_4 production. A modified version of the technique was used by Harper *et al.* (2004), who measured the total volume of gases (e.g. NH_3), and percentage composition of each gas over time, from submerged open-bottom carboys. The simplicity of the approach is a definite advantage; however, concerns remain as with all chambers that modify the temperature and airflow regimes normally associated with open lagoons.

Tracer release techniques

The emission of a gas can be determined by relating a target gas of interest to a second gas (tracer) with a known release rate and a co-located concentration measurement of the gases. The underlying assumption is that the transport of the tracer and targeted gas is the same along the common dispersion path. As a result, the relationship (Eqn 15.3) of the emission-to-concentration ratio (where the concentration terms are corrected for background concentration) along the dispersion path is identical for both gases. Therefore, knowing the target and tracer gas concentrations (C_G and C_T, respectively) and the tracer release rate (F_T), the gas emission (F_G) can be calculated as:

$$\frac{F_G}{C_G} = \frac{F_T}{C_T} \text{ rearranging } F_G = F_T \frac{C_G}{C_T} \qquad (15.3)$$

This technique has been applied to individual ruminants to measure CH_4 emissions associated with enteric fermentation (enteric tracer technique) where the tracer is released in the rumen. A second approach is to release a tracer gas into the air in proximity to the source of the emission (atmospheric tracer technique). With few exceptions, the tracer gas commonly used is sulfur hexafluoride (SF_6). This is a straightforward technique but assumes dispersion is the same between gases.

Enteric tracer technique

The most widely utilized tracer technique for determining enteric CH_4 emissions in ruminants was developed by Johnson *et al.* (1994) where the tracer gas is SF_6. The SF_6 is released at a constant rate (1–2 mg day^{-1} for sheep; 2–7 mg day^{-1} for cattle) from a permeation tube placed in the rumen (reticulo-rumen) of the animal. Prior to placement in the rumen, the release rate of SF_6 is determined by weighing the tubes (stored at 39°C) over several weeks. In selecting permeation tubes for an animal trial, care is needed to ensure the release rates are similar since it has been reported that the rate of release of SF_6 can influence the calculated CH_4 emission (Vlaming *et al.*, 2005), thus increasing the between-animal variability.

A continuous sample of expired and eructated air collected over typically 24 h near the nostrils and mouth is stored in an evacuated canister mounted on the animal. After removal of the canister from the animal, it is pressurized with nitrogen and then analysed for CH_4 and SF_6 concentrations using gas chromatography (GC). Background concentration of CH_4 and SF_6 tracer are subtracted to obtain the rumen-derived concentrations. A detailed protocol for the SF_6 tracer technique is reported by Johnson *et al.* (2007).

There has been concern expressed about the assumption of a constant SF_6 release rate from the permeation tube located in the rumen of the animal (Ulyatt *et al.*, 1999). The root of the issue is that the SF_6 tracer technique cannot be calibrated since the permeation tube cannot be recovered except where a rumen cannula is used. Lassey *et al.* (2001) reported an error of up to 15% in CH_4 emission associated with the assumption of a constant release rate, and offered a means of correcting this assumption.

The use of the SF_6 tracer technique inside barns with low ventilation may also be problematic since this can lead to a high background concentration of SF_6 that approaches the sampled SF_6 concentration at the nostrils/mouth. Very small differences (measured in parts per trillion, ppt) between the background and canister SF_6 concentrations can approach the resolution of detection, and cause serious error in the CH_4 emission calculation. The use of the SF_6 tracer technique on animals with a poorly fitted rumen cannula can contribute to elevated SF_6 concentrations (Beauchemin *et al.*, 2012). In this case, the assumption that all the SF_6 exits the rumen via the same path and in the same proportion as the CH_4 cannot be assumed.

Another limitation of this technique is related to its application for evaluating treatment differences. Since the technique uses the animal as the measuring unit, the technique error as well as the within- and between-animal variability must be well understood before considering using the SF_6 tracer technique for treatment differences. Despite these limitations, earlier work by Ulyatt *et al.* (1999) reported that with sufficient animal numbers, the SF_6 tracer technique gave similar emissions to other emissions techniques. They reported that the technique is especially useful for grazing CH_4 measurements, but that there is a need to know the dry matter intake accompanying the CH_4 emissions, since intake affects CH_4 emissions and confounds treatment differences.

Atmospheric tracer technique

A tracer gas can be released into the atmosphere external to the animal to measure CH_4 emissions (Eqn 15.3). The assumption in this case is that the tracer and targeted gas (CH_4) are mixed in the air by turbulence diffusion, in the same fashion. Johnson *et al.* (2002) released SF_6 as a tracer (at 50.2 μg h^{-1}) in a room housing cattle and the concentrations of CH_4 and SF_6 were measured once the concentrations reached steady-state. The concentrations inside the room were measured from collected air samples using GC. Kaharabata *et al.* (2000) released SF_6 at 16 locations (*c.*20–32 g h^{-1} per location) inside a barn containing 90 dairy cows, and measured CH_4 and SF_6 concentrations downwind (of the barn) by analysing air samples using GC. Griffith *et al.* (2008) used a controlled release of nitrous oxide (N_2O) as a tracer gas from points along the upwind side of a field containing grazing cattle. The average CH_4 emission from the small confined herd was accomplished by measuring N_2O and CH_4 concentrations simultaneously downwind of the confined herd using a FTIR (Fourier Transform InfraRed) open-path sensor. In this latter study, there is a clear advantage to using a tracer gas that is compatible with an analyser that allows continuous monitoring.

The use of a tracer like SF_6 allows the calculation of the herd's average CH_4 emission rate continuously throughout the day. Although the calculations are straightforward based on Eqn 15.3, it requires the use of a technically demanding (and expensive) sensor to monitor multiple diluted gases simultaneously.

Micrometeorological techniques

Micrometeorological techniques rely on the estimation of the ventilation rate of the air affected by the source. Unlike the chamber technique (FT-SS design) where the ventilation rate of the chamber is relatively easy to measure inside ducts, and in the trace technique where it is characterized using a tracer gas, the micrometeorological technique requires that the ventilation be estimated from wind measurements. The main advantage of micrometeorological techniques is that they are relatively less intrusive compared with chambers that modify the environment, or the enteric SF_6 tracer technique that requires that the animals be handled daily.

Eddy covariance (EC)

The EC technique has been used for monitoring emissions of CH_4 from grazing livestock (Dengel *et al.*, 2011), and has potential for determining NH_3 and CH_4 losses from large uniform manure storage facilities. It requires fast-response sensing of the vertical wind speed and associated gas concentration. Essentially the emission (F_G; g s^{-1}) is the cross-product of the instantaneous vertical wind speed (w'; m s^{-1}), measured with a sonic anemometer, and the instantaneous gas concentration (C'_G; gm^{-3}), usually averaged over a period of 10–30 min ($F_G = \overline{w'C'_G}$). The technique is technically demanding since it requires fast-response sensors (typically at 10 Hz) to capture the characteristic of the vertical eddies. It also requires a large homogeneous manure source to create a thick internal boundary layer. Because of these requirements, the EC technique has had a limited application to CH_4 emissions from livestock, and CH_4 and NH_3 from livestock manure storage facilities. It would be more applicable to nitrogen-amended soils, as used by Famulari *et al.* (2004).

A modification of the EC technique is the relaxed eddy accumulation (REA) technique, where the gas sampled at an inlet is partitioned into two streams corresponding to when the vertical wind is either moving upwards or downwards. This requires quick response switching of air streams in order to isolate the upward and downward winds. The gas in each of two air streams is accumulated and the emission (F_G; g m^{-2} s^{-1}) is determined as $F_G = \beta\rho\sigma_W\left(\overline{\chi_U} - \overline{\chi_D}\right)$ (from Baum and Ham, 2009), where β is a dimensionless factor, ρ is the air density (g m^{-3}), σ_w is the standard deviation of the vertical wind speed (measured with a sonic anemometer; m s^{-1}), and χ is the mixing ratio (e.g. ppb) of the upward (U) and downward (D) moving gas.

Flux gradient (FG)

The FG technique uses measurements of gas concentration at two heights made within the internal boundary layer – the layer of air above the surface that characterizes the energy, mass and momentum exchanges at the surface. The emission from the source is the product of concentration gradient $\left(\Delta C / \Delta z\right)$ and a transfer coefficient, known as the eddy diffusivity (K), given in the relationship:

$$F_G = \frac{-k(z_1 - d)(z_2 - d)}{S_c\phi_m^2(z_1 - z_2)^2}\Delta C\,\Delta u \qquad (15.4)$$

where k is a constant (0.4; Von Karmen), z_1 (lowest) and z_2 are the heights (m) above the surface, d is the zero plane displacement (m) that is determined using data collected with a three-dimensional anemometer, S_c is the Schmidt number assigned a value of 0.63 (Flesch *et al.*, 2002), $Ø_m^2$ is the non-dimensional correction for the effect of thermal stability on the wind profile, ΔC is the gas concentration (µg m^{-3}) difference between z_2 and z_1, and Δu is the horizontal wind speed (m s^{-1}) difference between z_2 and z_1. The value of $Ø_m$ is calculated as a function of the atmospheric stability (stable or unstable) and is given as:

$$\phi_m = \left(1 - 19\frac{z}{L}\right)^{-0.25} \text{ for unstable conditions} \qquad (15.5)$$

$$\phi_m = 1 + 5.3\frac{z}{L} \text{ for stable conditions} \qquad (15.6)$$

where z/L is the Monin–Obukhov parameter and L (Monin–Obukhov length) can be determined using the sensible heat flux (H) and friction velocity (u_*) as derived from the output from a three-dimensional anemometer at height z ($L<0$ unstable conditions; $L>0$ stable conditions).

The FG technique has been used to estimate enteric CH_4 emissions from livestock by Judd *et al.* (1999) and Laubach *et al.* (2008), and NH_3 from beef feedlot manure by Todd *et al.* (2005).

Mass balance (IHF)

In a similar way that the difference in mass of the gas entering and leaving a chamber is used to calculate (along with the airflow in the ducts) the emission from the enclosed source, so too can the emission from a source in the open (non-enclosure) be estimated. In this case, the incoming and outgoing air over the source is characterized by dividing it into layers, where the wind speed and gas concentration is measured in each layer. The difference between incoming and outgoing gas concentration in each layer is then multiplied by the corresponding wind speed – this horizontal flux is then summed for all layers to yield the integrated horizontal flux (IHF):

$$F_G = \frac{1}{x}\sum_{z1}^{z2}\left[\bar{u}\left(\bar{C}_O - \bar{C}_I\right)\right]\Delta z \qquad (15.7)$$

where x (m) is the distance between the incoming and outgoing perimeters (fetch), $\bar{u}$ is the time-averaged wind speed (m s^{-1}; at mid-point of a perimeter) in a layer, and $\bar{C}_O$ and $\bar{C}_I$ are the outgoing and incoming time-averaged concentration (g m^{-3}; at a single point mid-way along the perimeter).

Typically, at least four measurement heights are used covering the complete internal boundary layer (the lowest height where C_O is equal to C_I). The IHF technique was employed by Laubach *et al.* (2008) to measure CH_4 emission from a small number of penned cattle. This technique was used for CH_4 emissions from circular (*c.*11 m diameter) open-top tanks filled with dairy manure (Vanderzaag *et al.*, 2011). However, for larger area sources, with a high internal boundary layer, it may not be feasible to measure over the complete internal boundary layer. In this case, Ryden and McNeill (1984) regressed wind speed and concentration against ln(z) and expressed F_G as a function of the regression coefficients.

An alternative approach to the IHF technique is to use a line-averaged concentration along each perimeter (not just at a single point), known as the mass difference (MD) technique. This technique was used by Denmead *et al.* (1998) and Harper *et al.* (1999) as a direct measure of CH_4 from confined cattle. The calculation of F_G is as follows:

$$F_G = X\sum_{z1}^{z2}\left[\bar{U}\left(\bar{C}_4 - \bar{C}_2\right) + \bar{V}\left(\bar{C}_3 - \bar{C}_1\right)\right]\Delta z \qquad (15.8)$$

where X is the perimeter length (m) of the square source area, $\bar{U}$ and $\bar{V}$ are the wind speeds (m s^{-1}) perpendicular to the perimeter orientation, and $\bar{C}_4$ and $\bar{C}_3$ are the line-averaged concentrations (g m^{-3}) along the downwind perimeters, while $\bar{C}_2$ and $\bar{C}_1$ the concentrations along the upwind perimeters.

The advantage of the mass balance (MB) technique is that it is simple to set up and does not require a great deal of expensive instrumentation. For example, acid traps can be used for NH_3 sampling along with cup anemometers to yield net NH_3 emission over an extended time frame (e.g. 24 h). This technique as well does not require atmospheric stability corrections. One disadvantage of the technique is the need to use multiple sampling measurements to correctly characterize the vertical concentration and wind speed profiles associated with the source.

Energy balance (EB)

The energy balance approach makes use of the concept that the transfer of a gas is equated to that for water vapour and heat in a specific layer above a uniform surface (within the internal boundary layer). Denmead *et al.* (1974) determined this transfer coefficient (h; m s^{-1}) using the relationship

$$h = \frac{(R - G)}{\rho C_p \Delta T_e} \qquad (15.9)$$

where R is the net radiation at the surface (W m^{-2}), G is the soil heat exchange (W m^{-2}), ρ is the air density (kg m^{-3}), C_p is the specific heat of air (kJ kg^{-1} °C^{-1}) and ΔT_e is the equivalent temperature gradient (°C; equivalent temperature is

corrected for adiabatic process) over the layer. The emission F_G is simply the product $h \times \Delta C_G$, where the latter term is the concentration gradient of the gas over the specific layer.

There has been little use of this technique for NH_3 or CH_4 since the study by Denmead *et al.* (1974) that focused on NH_3 emissions from a 4-ha pasture with grazing sheep. A major reason this has received little attention as a technique for enteric CH_4 (e.g. associated with a grazing or confined herd), or for CH_4 or NH_3 from manure storage, is the need for a large and uniform source.

Theoretical profile shape (TPS)

The theoretical profile shape (Wilson *et al.*, 1982) specifies that there is a unique height above an emitting homogeneous surface, referred to as height ZINST, where the vertical emission from the surface is proportional to the horizontal flux. The value of ZINST is determined using a dispersion model or from the curves derived by Wilson *et al.* (1982). The technique has the advantage that a single concentration and wind speed measurement, usually located in the centre of plot, can be used to estimate emissions. The technique has been mainly used to estimate NH_3 emissions from land-applied manure (Gordon *et al.*, 1988). For the most part, this technique has been superseded by developments in dispersion modelling that allow measurements to be made anywhere in the emitting plume.

Modelling dispersion (BLS)

The gas emitted by a source in the open air moves in three dimensions, along the wind direction axis, and vertically and horizontally across wind directions. Thus, the volume of the plume 'grows' with distance from the source, causing a dilution effect on the concentration of the emitted gas at the source. The rate of growth (or the dispersion) of the gas plume is determined from the mixing capacity of the air flowing over the source. Much work has been conducted in micrometeorology to estimate the boundary layer mixing capacity based on field measurements of wind speed (mechanical mixing) and temperature (buoyancy effect).

One approach to dispersion modelling that has gained wide usage is the backward-time Lagrangian stochastic (BLS) dispersion model (Flesch *et al.*, 1995). Essentially, the relationship of F_G at the source to concentration (C) anywhere in the plume is calculated using a dispersion model that is driven by measurements of wind statistics and the mixing capacity of air. The predicted F_G is the product of the simulated ratio $(F_G/C)_{SIM}$ and the measured gas concentration. The simulated ratio is generated by the model that tracks the movement of gas particles (*c.*50,000) from the location inside the plume where concentration is measured, to the upwind source itself (i.e. backwards in time). This backward approach saves computation time whereas if the trajectories of particles were to be computed from the source (forward approach), many of the trajectories would miss the position where the gas concentration was monitored and therefore would not be used in the computation.

The choice of location where concentration measurement is made in the dispersing gas plume is critical to the success of this technique. Locating the sensing unit too far from source, where dilution of the gas approaches the background concentration, will reduce the accuracy of the F_G calculation. Conversely, measuring too close to the source causes error in the F_G term since the assumption of homogeneous dispersion is compromised by non-uniform turbulence around the source. For large uniform sources, it is possible to position the concentration and wind instruments above the source at a central location. This configuration removes the impact of wind direction, i.e. a more continuous time series can be collected since the source is always upwind.

The BLS technique has proven useful in estimating the NH_3 and/or CH_4 from open feedlots (Flesch *et al.*, 2007; Haarlem *et al.*, 2008; McGinn *et al.*, 2008a). It has also been used at large dairy lagoons by Todd *et al.* (2011) to measure CH_4 emissions, and by McGinn *et al.* (2008b) for NH_3 emissions. The major advantage of this technique is that, like all micrometeorological techniques, it is non-intrusive and may provide continuous measurements needed to determine emissions. Harper *et al.* (2009) reported, on average, that the BLS technique estimated 100.1% recovery rates from known true release rates. The BLS technique was shown

(McGinn *et al.*, 2009) to be sufficiently precise to allow evaluation of the differences in CH_4 emission from cattle under different dietary treatments. There are also disadvantages, which include the need for expensive instrumentation to measure: (i) wind statistics needed to estimate atmospheric stability; and (ii) background concentration and concentration downwind from the source. Another limitation of the BLS technique is the loss of emission data when atmospheric conditions are not conducive to the assumptions in the BLS model (Flesch *et al.*, 2007).

Comparison of Techniques

Differences in emissions between treatments can result from instrumentation error in making the emission measurements, and error in the emission technique (e.g. invalid assumptions). It is recommended where possible that more than one technique be used to evaluate emissions in order to develop an appreciation for variability attributed to the measurement system. Alternatively, comparison of emission data by different studies may aid in understanding variability due to technique, but clearly these comparisons will be confounded by differences in the studies such as animal and diet factors. For example, Boadi *et al.* (2004) compared CH_4 emissions from dairy cows in different studies that also used different techniques and reported a wide range in CH_4 emissions varying from 286 to 433 g per animal per day.

Methane emissions for beef cattle and sheep have been compared using different techniques within a single study (Table 15.1). The SF_6 tracer technique was reported to be 93–95% of that measured using whole-animal chambers (Johnson *et al.*, 1994; Ulyatt *et al.*, 1999; McGinn, 2006) and 105% of that measured using hood chambers (Boadi *et al.*, 2002). Grainger *et al.* (2007) found the SF_6 tracer technique to overestimate the chamber technique by 2%. Pinares-Patiño *et al.* (2008) also compared the two techniques, and found that the SF_6 tracer technique (coefficient of variation, CV, ranged from 7.8 to 18.4) was more variable compared with the chamber technique (CV 4.3–7.7). The higher CV of the SF_6 tracer technique was inferred to be caused by a change in the SF_6 tracer release rate.

For the BLS dispersion technique, McGinn *et al.* (2009) found that the BLS underestimated CH_4 emissions from penned cattle by 7% relative to the SF_6 tracer technique. In this application, the gas plumes from the individual animals were estimated, and the animal position was measured using global positioning devices. Laubach *et al.* (2008) reported a good fit of CH_4 emissions from confined cattle over a period of days between the BLS, FG and IHF techniques, with the provision that a separation distance of cattle and sensors be at least 20 m for the FG and IHF techniques. In this application, the emissions were treated as a uniform area source.

A good fit of emission data existed using the BLS dispersion and the IHF techniques for NH_3 emissions from stocked beef cattle manure (Sommer *et al.*, 2004). However, the use of static chambers (NFT-NSS) was shown in this same study to underestimate greatly the emissions of several gases including CH_4 by at least 78% relative to the micrometeorological techniques. The cause of this underestimation may be related to obstruction to airflow from the pile, which vented in a similar fashion to a series of chimneys.

Shah *et al.* (2006) reported on several attributes of enclosures and three micrometeorological techniques, the IHF, FG and EC, all used to estimate NH_3 emissions. They rated enclosures to have low-medium reliability while the IHF had high reliability (FG and EC not rated on this attribute).

Conclusions

Before initiating any emission measurement study, the possible quantitative measurement systems need to be scrutinized to avoid known limitations associated with each technique. This involves understanding the assumptions underlying the techniques, and the advantages and disadvantages for a given technique within the context of the study parameters. Of key consideration is a needed knowledge of the accuracy and precision of each technique to ensure the results will be applicable to the treatments (usually looking at mitigation strategies) being used in mitigation studies. In this case, the precision must be greater than the differences in emissions imposed by the treatments.

For measuring CH_4 emissions from ruminants, the use of FT-SS chambers has seen wide adoption since they can be easily calibrated and offer a sensitive measure to emissions attributed to dietary differences. In this environment the differences in feed intake can also be easily monitored and removed from the treatment effect. The SF_6 tracer technique is commonly used for enteric CH_4 emission measurements as well, but it is generally accepted to yield greater variability and hence not as sensitive to treatment differences as the FT-SS chamber technique. When the SF_6 tracer technique is used in a grazing system, determining the feed intake (not easily measured) also confounds treatment differences. Micrometeorological techniques for measuring enteric CH_4 emissions from ruminants have more of a role in developing emission factors for different livestock since many of these techniques require a large scale, e.g. farm or herd, and therefore replication needed for mitigation purposes is more problematic.

In measuring CH_4 and/or NH_3 from manure, often the outdoor environment is critical to the emission. In this situation, the micrometeorological techniques prove useful since these techniques are not intrusive. Surface chamber techniques suffer from spatial sampling variability and typically do not provide a continuous measure, whereas micrometeorological techniques are more conducive to continuous measurements.

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16 Manipulation of Microbial Ecology for Sustainable Animal Production

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Introduction

The world's human population will grow from almost 7 billion people to over 9 billion in 2050. The projections show that feeding a population of 9 billion people would require increasing overall food production by some 70% between now and 2050 (FAO, 2009). Accordingly, production in the developing countries would need almost to double (FAO, 2009). As the world's population increases, so do problems associated with food security, land use and social and environmental issues (Parker, 2011). In the past, increases in the demand for food were achieved by bringing more land into production, but the amount of land required to do so is not realistic given the projected trends in population growth (Smith *et al.*, 2010). For example, it has been estimated that the availability of new arable land necessary to provide food for the world's growing population would require the clearing (i.e. deforestation) of 900 million additional hectares of land (e.g. nearly one-third of the contiguous USA) just to maintain the current caloric levels (Biello, 2011), or 2.1 billion additional acres (e.g. about two-thirds of the contiguous USA) to meet the necessary caloric levels of the world's population. Yet, land area is less of a limitation than the availability of water.

As we look for opportunities to feed a growing population, waste and inefficiencies in the current agricultural systems will not be acceptable in 2050. Furthermore, current agricultural and food production practices contribute significantly to the global problem of greenhouse gas emissions. Therefore, a multifunctional approach to agriculture will be required to significantly increase food production sustainably using the existing land area and water resources. Broadly speaking, multifunctional agriculture means that agricultural activity, beyond its role of producing food and fibre, may also have several other roles such as renewable natural resources management, landscape and biodiversity conservation and contribution to the socio-economic viability of rural areas (Renting *et al.*, 2009).

The increasing demand for food from animal sources, especially in the developing countries, will lead to a continued high interest in the interaction of livestock with the environment. About 25% of the world's land is used exclusively for grazing livestock, accounting for nearly 30% of the cattle and 35% of the sheep and goats (Seré *et al.*, 1996). Furthermore, most of the rest of the world's approximate 1.1 billion ruminants (FAO, 2000) are reared in mixed farming systems where grazing is complemented with

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crop by-products and stubble (Masters and Wright, 2005).

Grazing livestock are significant contributors to anthropogenic greenhouse gas production, particularly through the production and release of methane gas. Methane acts as a potent greenhouse gas with 25 times more global warming potential than carbon dioxide. Methane is an end-product of enteric fermentation of plant material by the complex microbial ecosystem that resides in the rumen and is released into the environment, mostly through eructation (Murray *et al.*, 1976). Enteric methane is produced in the rumen when hydrogen is released by other microorganisms (e.g. fungi, rumen protozoa) during fermentation and used by methanogenic archaea, also called methanogens, to reduce carbon dioxide. Thus, methane is produced by the microorganisms living inside the rumen rather than by the animal itself.

Domesticated ruminants (cattle, goats, sheep and water buffalo) produce as much as 86 million metric tons of methane per year (McMichael *et al.*, 2007), of which, approximately 55.9 million metric tons of enteric methane are from non-dairy cattle, 18.9 million metric tons are from dairy cattle, 9.5 million metric tons are from sheep and goats, and 6.2–8.1 million metric tons are from water buffalo (Johnson and Ward, 1996; McMichael *et al.*, 2007). In comparison, the global yearly enteric methane contribution from non-ruminant livestock has been estimated to be 0.9–1.1 million metric tons from camels and pigs and 1.7 million metric tons from horses (Johnson and Ward, 1996). In many countries, ruminant livestock is the single largest source of methane emissions from the agricultural sector. In Australia and the USA, enteric methane accounts for approximately 70% and 73%, respectively, of agricultural methane emissions (Environmental Protection Agency, 2010; National Greenhouse Gas Inventory, 2010). This equates to nearly 55.6 million metric tons carbon dioxide (CO_2) equivalents of methane from Australian livestock and nearly 141 million metric tons CO_2 equivalents of methane from US livestock. This represents a significant loss of feed energy from animal agriculture and an economic loss to farmers because the feed is converted to methane instead of production outputs, such as wool, beef or milk.

Because the continuous growth of the human population is expected to result in an increase in the number of domesticated ruminants, high methane production from domesticated ruminant livestock is an important undesirable trait due to its negative impact on animal production and its contribution to climate change. Thus, reducing methane emissions by livestock has become an integral part of sustainable agriculture and climate change abatement strategies (Thorpe, 2008). Opportunities to manipulate gas production, either per animal or per unit of product, through feeding strategies and rumen manipulation, are now being identified.

Rumen Microbiome

The microbial ecosystem is very complex and involves thousands of species of bacteria (10^{10}–10^{11} cells ml^{-1}), archaea (10^7–10^9 cells ml^{-1}), protozoa (10^4–10^6 cells ml^{-1}), fungi (10^3–10^6 cells ml^{-1}) and viruses (10^9–10^{10} cells ml^{-1}) (Klieve and Swain, 1993; Koike and Kobayashi, 2009), which play an important role in the digestion of feed, and interact with their host and each other. The rumen microorganisms also supply energy and protein to the host in the form of volatile fatty acids and microbial protein (Hungate, 1966). Not only is the ecosystem complex, but also relatively poorly understood, particularly inter-species interactions and interactions with the host. For that reason, when considering methane mitigation strategies, the relationship that rumen methanogens have with other microorganisms is especially important. For the context of this chapter, discussion will focus on targeting the methane-producing methanogens and or decreasing methanogenesis.

Methanogenic archaea

There are over 120 species of methanogens representing seven orders, which include 13 families and 35 genera (Euzéby, 2012). For a recent review of the diversity of gut methanogens in herbivorous animals, see St-Pierre and Wright (2012). Most methanogens (e.g. *Methanobrevibacter, Methanobacterium, Methanomicrobium*) are able to use hydrogen, carbon dioxide and formate as

substrates for methane production, but some methanogens (e.g. *Methanosarcina barkeri*) are also able to produce methane from acetate, methylamines and methanol (Stewart *et al.*, 1997).

Until recently, our understanding of the diversity and microbial interactions between methanogens and other rumen microorganisms has been very limited. However, knowledge of this ecosystem is rapidly accumulating, particularly with the advances of molecular biology (e.g. bar-coded pyrosequencing) and culture independent technologies. Molecular-based studies from Australia, New Zealand and the USA have made significant contributions to our knowledge of methanogen ecology and diversity in the rumen (Miller and Wolin, 1986; Lin *et al.*, 1997; Sharp *et al.*, 1998; Tokura *et al.*, 1999; Jarvis *et al.*, 2000; Yanagita *et al.*, 2000; Tajima *et al.*, 2001; Whitford *et al.*, 2001; Irbis and Ushida, 2004; Shin *et al.*, 2004; Skillman *et al.*, 2004, 2006; Wright *et al.*, 2004b, 2006, 2007, 2008; Janssen and Kirs, 2008; Ouwerkerk *et al.*, 2008; Brulc *et al.*, 2009; Hook *et al.*, 2009, 2011; Sundset *et al.*, 2009a, b; Williams *et al.*, 2009; Zhou *et al.*, 2009, 2010; Leahy *et al.*, 2010; Pei *et al.*, 2010; King *et al.*, 2011; Franzolin *et al.*, 2012; Lee *et al.*, 2012). Many of these studies have indicated that gut methanogens tend to differ depending on diet and geographical location, and that *Methanobrevibacter* phylotypes are the dominant methanogens in ruminant livestock worldwide (Wright *et al.*, 2008).

Rumen protozoa

Rumen ciliates (Fig. 16.1) are involved in host metabolism and digestion of plant material in the rumen. Young ruminants isolated at birth do not contain rumen protozoa (Eadie, 1962; Dehority, 1978; Fonty *et al.*, 1988), but become faunated as a result of adults regurgitating food and rumen contents back into the mouth during rumination and salivating on feed, which is then consumed by the young animal, or the protozoa are passed directly by the mother to its offspring during grooming (Dehority, 1993).

There is a complex relationship between the ciliated protozoa (e.g. ciliates) and methanogens in the rumen. Rumen protozoa are one of the major sources of hydrogen in the rumen, and methanogens living on (extracellular) and within (intracellular) the rumen ciliates may generate up to 37% of the methane emissions from ruminant animals (Finlay *et al.*, 1994; Hegarty, 1999a, b). Accordingly, when ruminants were defaunated (i.e. made protozoa-free), they produced on average 13% less methane than faunated ruminants (Kreuzer, 1986; Hegarty, 1999a). Commonly observed protozoa in the bovine rumen that have this unique association with the methanogens are from the genera *Entodinium*, *Epidinium*, *Ophryoscolex* and *Polyplastron*, while, the methanogens most often associated with the protozoa are from the orders *Methanobacteriales* and *Methanomicrobiales* (Sharp *et al.*, 1998).

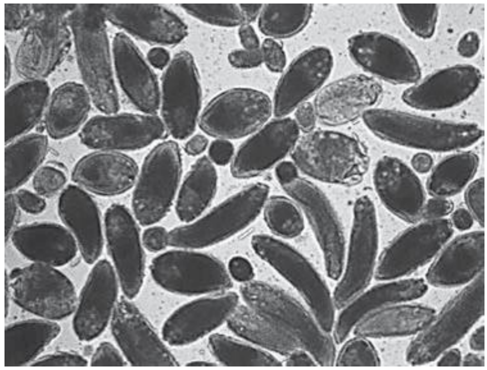

Fig. 16.1. A photomicrograph of a mixed population of rumen protozoa.

Rumen protozoa have been estimated to account for half of the microbial biomass in the rumen, and contribute to up to one-third of fibre digestion (Hungate, 1966; Williams and Coleman, 1997). The rumen ciliates *Isotricha* and *Dasytricha* (Fig. 16.2) are very important in utilizing soluble sugars and controlling the rate of carbohydrate fermentation, especially when large quantities of soluble carbohydrates are present in the diet, whereas some entodiniomorphid ciliates are responsible for controlling starch digestion by engulfing whole starch granules. However, some rumen ciliates can also have a negative impact on ruminant protein metabolism. Microbial protein accounts for as much as 90% of the amino acids reaching the small intestine of the animal. The rumen ciliates, also a source of microbial protein, do not

Fig. 16.2. A photomicrograph of *Isotricha prostoma* (160 × 90 μm) and the smaller *Dasytricha ruminantium* (58 × 27 μm).

appear to pass as rapidly to the small intestine. Rumen ciliates also consume bacterial protein that could otherwise be used by the animal and are only able to convert about 50% of bacterial nitrogen to protozoal protein (Coleman, 1975). In addition, the larger ciliates consume smaller ciliates. As a result, the total microbial protein flow to the small intestine is generally reduced and excess rumen ammonia is increased in faunated animals (Bird and Leng, 1978).

Methane Reduction Strategies

For intensive agriculture to become a sustainable industry with minimal environmental impact, understanding rumen microbial ecology is critically important. For many years, researchers have tried to manipulate the numbers or activity of the rumen methanogens in order to improve the efficiency of ruminant production in an ecologically sustainable way. A number of mitigation strategies for decreasing enteric methane emissions from grazing ruminants have been suggested, such as dietary composition, increasing the synthesis of propionate and long chain fatty acids in the rumen to minimize methane production, defaunating agents, mononesin, propionate enhancers, nitrocompounds, ionophores, halogenated methane analogues, genomics, biological control and immunization, to name just a few (Van Nevel and Demeyer, 1995; Czerkawski *et al.*, 1966; Mbanzamihigo *et al.*, 1996; McCrabb *et al.*, 1997; Mathison *et al.*, 1998; Hegarty, 1999a, b; Joblin, 1999; Klieve and Hegarty, 1999; Machmüller and Kreuzer, 1999; Anderson *et al.*, 2003; Machmüller *et al.*, 2003a, b; Wright *et al.*, 2004a; Denman *et al.*, 2007; Attwood and McSweeney, 2008; Mitsumori and Sun, 2008; Williams *et al.*, 2008). Currently, the unique features of rumen methanogens are being characterized using comparative genomics, and this new information is providing researchers with a host of options for anti-methanogen strategies (Leahy *et al.*, 2010).

Productivity may also be enhanced through the use of production enhancing agents such as bovine somatotropin and anabolic steroids, for which emission reductions up to 9% have been demonstrated (Johnson *et al.*, 1992). Because of the growing concern over the use of antibiotics and chemicals in animals used for human consumption, the ideal mitigation strategies will have to be safe, leave no residues in meat and milk, be cost effective, practical, applicable to grazing animals and suitable for on-farm application. However, if sustained reductions in methane emissions do not lead to increased milk and meat production through improved animal performance, then adoption or acceptance of these strategies will be non-existent. Thus, incentives or support may be needed to encourage the uptake of methane-reduction strategies.

Dietary composition

A relationship exists between nutritional status, productivity and methane emissions (Leng, 1992). The amount of methane generated in the rumen is influenced by dietary factors. Assuming reasonable feed quality, feed intake by itself has a major impact on methane production. Simply stated, increasing the nutritional quality of forages is one way of lowering methane emissions, as animals grazing high-quality (i.e. high organic matter digestibility) pastures will produce the least amount of methane per unit of intake. A higher-quality diet increases intake and the rate of live weight gain, therefore decreasing the amount of methane emitted per unit of production (i.e. weight gain or milk production), despite the fact that methane emitted

per day from fermentation of a given amount of feed increases slightly as digestibility increases (Hegarty, 1999b). Conversely, a poor-quality feed produces more enteric methane. Although these are very basic influences, feed intake and quality do affect microbial communities in the rumen, and although not well understood, advancements in this area could lead to greater reductions in the future.

The components of the ruminant diet are important for methane production as they are able to influence the rumen pH and subsequently alter the microbiota present (Johnson and Johnson, 1995). The digestibility of cellulose and hemicellulose is strongly related to methane production. In a study by Holter and Young (1992), a positive relationship was found between digestibility of hemicellulose and methane output in forage fed non-lactating cows. However, a negative relationship was found between digestibility of cellulose and methane output (Holter and Young, 1992). The relationship between methane production and percentage of concentrate in the diet is curvilinear, with methane losses of 6–7% of gross energy being constant at 30–40% concentrate levels in the diet and then decreasing to 2–3% of gross energy with a concentrate proportion of 80–90% (Sauvant and Giger-Reverdin, 2007).

The starch component of the diet is also known to promote propionate formation, through a shift to amylolytic bacteria and a reduction in rumen pH, leading to a decrease in methanogenesis (Van Kessel and Russell, 1996). Grinding forage feed before it is ingested by the cows also seems to decrease the production of methane, presumably by increasing the rate of digestion and flow through the gastrointestinal tract, thereby limiting the time available for methane to be produced within the rumen (Johnson and Johnson, 1995). It is important to note that increasing the amount of rapidly fermentable carbohydrates in a diet can increase the rate of passage from the rumen, as well as lower rumen pH.

Defaunation

Defaunation, the removal of ciliated protozoa from the rumen, has been examined as a way to increase protein outflow to the small intestine and therefore animal production (Bird and Leng, 1984; Jouany *et al.*, 1988; Williams and Coleman, 1992). In previous studies, defaunation of the rumen in sheep has produced 7–53% more wool growth under a range of dietary conditions, such as pasture, cereal chaff or straws with protein and sugar supplements, and silage (Bird *et al.*, 1979, 1994; Bird and Leng, 1983, 1984; Fenn and Leng, 1989; Forster and Leng, 1989a, b; Habib *et al.*, 1989; Ivan *et al.*, 1992; Hegarty *et al.*, 2000).

It has been estimated that methanogens associated protozoa, both intracellularly and extracellularly, are responsible for 9–37% of the methane production in the rumen (Finlay *et al.*, 1994; Newbold *et al.*, 1995; Machmüller *et al.*, 2003b). For this reason, treatments that decrease the protozoal population of the rumen may also decrease the protozoa-associated methanogen population and therefore decrease the methane production within the rumen. Treatments normally used to defaunate the rumen include copper sulfate, acids, surface-active chemicals, triazine, lipids, tannins, ionophores and saponins (Hook *et al.*, 2010). However, maintenance of defaunated animals can be difficult.

Organic acids

Fumaric acid is a trans-butenedioic acid widely found in nature. Unfortunately, the *in vivo* effects of fumaric acid on methane abatement were found to be inconsistent. In one study (Wood *et al.*, 2009), 100 g kg^{-1} fumaric acid was offered to growing lambs, resulting in a 62–76% reduction in methane output. However, when it was fed to growing beef cattle, steers and wethers, it did not significantly decrease methane emissions.

Malic acid is a dicarboxylic acid, which is made by all living organisms, contributes to the pleasantly sour taste of fruits and is used as a food additive. When beef heifers were fed diets supplemented with 3.75% and 7.5% malic acid on a dry matter basis, methane was decreased 3% and 9%, respectively (Foley *et al.*, 2009). Although it appears that organic acids may provide some beneficial effects for methane

abatement, further studies are needed to determine the optimal conditions for use and if the benefits are lasting.

Monensin

Monensin is a polyether antibiotic, which is banned in the EU, but widely used in North America to increase feed efficiency, weight gain and milk production (Hook *et al.*, 2010). Monensin inhibits Gram-positive microorganisms and selects for Gram-negative microorganisms, which causes a shift towards propionate production in the rumen (Bergen and Bates, 1984; Russell and Strobel, 1989). It is believed that monensin indirectly decreases methane production by inhibiting the growth of the bacteria and protozoa (Bergen and Bates, 1984; Russell and Strobel, 1989).

Methane abatement from monensin supplementation is highly variable, ranging up to 25% with different outcomes for the duration of these effects (Hook *et al.*, 2010). It is believed that adaptation of the rumen microbiome to monensin may occur, thereby decreasing subsequent treatments of the antibiotic. In a study by Guan *et al.* (2006), steers were fed either a low-concentrate diet or a high-concentrate diet while supplemented with monensin. Methane output and rumen ciliate densities were decreased 27% and 77% respectively, over the initial 4 weeks on the low-concentrate diet; and 30% reduction in methane output and 83% reduction in the ciliate protozoal population over the intial 2 weeks on the high-concentrate diet. However, both methane levels and protozoal numbers returned to baseline within 4–6 weeks, probably because the rumen ciliate population adapted to monensin and returned to pre-treatment levels (Guan *et al.*, 2006). Although monensin has been shown effectively to reduce methane output in ruminants, there are differences in the degree of abatement depending on the diet and animal used (Odongo *et al.*, 2007a).

Lipids

Lipids, such as fatty acids and oils, are options for feed supplementation that have been investigated both *in vitro* and *in vivo* for their effects on methanogenesis (Hook *et al.*, 2011). There are many factors that may account for the varying effect of lipids on methane abatement, such as the ruminant species, experimental diet and the type of lipid used. By incorporating lipids in the feed, methane production is decreased, either by inhibiting the protozoa, or by increasing the production of propionic acid (Johnson and Johnson, 1995). Interestingly, Czerkawski *et al.* (1966) demonstrated that oleic acid (C18:1), linoleic acid (C18:2) and linolenic acid (C18:3), when individually infused into the rumen of sheep, decreased methane generation by 27%, 26% and 34%, respectively.

Fatty acids are thought to directly inhibit methanogens by binding to the cell membrane and interrupting membrane transport (Dohme *et al.*, 2001), whereas unsaturated fatty acids may be used as hydrogen acceptors as an alternative to the reduction of carbon dioxide (Johnson and Johnson, 1995). This has led to a number of fatty acids being investigated *in vivo* for methane suppressing effects. Myristic acid was found to decrease methane by 22% in sheep fed a forage-based diet, by 58% in a concentrate-based diet when 50 mg kg^{-1} dry matter was used (Machmüller *et al.*, 2003a) and by 36% methane (Odongo *et al.*, 2007b) in dairy cattle fed a total mixed ration with 5% myristic acid supplementation on a dry matter basis. Moreover, *in vitro* studies have found that fatty acids, used in combination, have the greatest suppression of methanogenesis due to a synergistic effect (Dohme *et al.*, 2001; Soliva *et al.*, 2004). Therefore, it is likely that oil supplementation would provide a more dramatic depression of methane production than individual fatty acids (Soliva *et al.*, 2004).

Soy oil, sunflower oil and palm kernel decreased methane production by 39%, 11.5–22.0% and 34%, respectively (Dohme *et al.*, 2000; McGinn *et al.*, 2004; Jordan *et al.*, 2006b; Beauchemin *et al.*, 2007a). However, coconut oil is the most popular oil for methane abatement (Hook *et al.*, 2011) and has been reported to decrease methane from 13% to 73%, depending on the inclusion level, diet and ruminant species used (Machmüller and Kreuzer, 1999; Machmüller *et al.*, 2000; Jordan *et al.*, 2006b). However, *in vivo* studies involving oil supplementation are often accompanied by a reduction in

dry matter intake, which also contributes to lower methane production (Machmüller *et al.*, 2000).

Tallow and whole soybeans have been shown to decrease methane by 11% and 25% per kg dry mater intake, respectively (Jordan *et al.*, 2006a; Beauchemin *et al.*, 2007a). However, palatability issues resulted in 60% refusals. Sunflower seed supplementation with heifers and dairy cows resulted in 23% and 10.4% reduction in methanogenesis, respectively (Beauchemin *et al.*, 2007a, 2009). Flaxseed and canola seed also decreased methane by 17.8% and 16.0% per kg dry matter intake, respectively (Beauchemin *et al.*, 2009). Supplementation with rapeseed, sunflower seed and linseed resulted in methane abatements of 19%, 27% and 10%, respectively, in growing lambs (Machmüller *et al.*, 2000). It is important to note that the majority of *in vivo* experiments undertaken to evaluate lipids have been short-term in duration. Long-term supplementation studies are needed comprehensively to assess the effectiveness of lipid supplementation as an abatement strategy. However, lipids can affect palatability, intake, animal performance and milk components, all of which can have implications for practical on-farm use (Jordan *et al.*, 2006a; Odongo *et al.*, 2007b).

Plant compounds

Saponins have been shown to inhibit protozoa and decrease hydrogen availability *in vitro* (Guo *et al.*, 2008). However, when Holtshausen *et al.* (2009) supplemented cows with whole-plant *Yucca schidigera* powder 10 g kg^{-1} dry matter or whole-plant *Quillaja saponaria* powder at 10 g kg^{-1} dry matter, no effect of the plant supplementation was found due to reduced feed digestion and fermentation *in vivo*, when compared to the *in vitro* trials.

Condensed tannins are thought to directly inhibit methanogens and limit methanogenesis by decreasing hydrogen availability (Carulla *et al.*, 2005; Puchala *et al.*, 2005; Tavendale *et al.*, 2005). *Lespedeza cuneata, Acacia mearnsii, Callinada calothyrsus* and *Fleminga macrophylla* all contain condensed tannins, and were found to make significant reductions (13–24%) in methanogenesis *in vivo* (Carulla *et al.*, 2005; Tiemann *et al.*, 2008). However, condensed tannin from *Schinopsis quebrachocolorado* (Beauchemin *et al.*, 2007b) and tannin-containing sorghum silage (De Oliveira *et al.*, 2007) fed to cattle did not suppress methanogenesis.

Essential oils have antimicrobial activities that inhibit Gram-positive bacteria (Burt, 2004; Calsamiglia *et al.*, 2007), which should reduce the amount of available hydrogen for methanogenesis. More *in vivo* research is needed with essential oils, condensed tannins and saponins to determine the optimal dosage where methanogenesis is reduced without producing negative side effects on digestibility (Hook *et al.*, 2010), or detection of residues in meat or milk (Calsamiglia *et al.*, 2007).

Immunization

A novel immunization approach was explored and tested to increase the efficiency of nutrient utilization in farmed ruminants and reduce methane emissions (Wright *et al.*, 2004a; Williams *et al.*, 2008, 2009). The idea was to stimulate the host ruminant's immune system to elicit an immune response and produce antibodies against the rumen methanogens. Wright *et al.* (2004a) vaccinated sheep with an anti-methanogen vaccine that was based on three strains (1Y, AK87 and ZA-10) belonging to the genus *Methanobrevibacter*. Vaccination induced a humoral immune response, as indicated by the specific Immunoglobulin G (IgG) titres in plasma and saliva, and specific anti-methanogen IgG was also delivered to the rumen as indicated by the titres in the rumen fluid. As a result, vaccination produced a 7.7% decrease in methane production per kg of dry matter intake.

Curiously, Wright and his colleagues (2006) later discovered that less than 20% of the methanogen species identified in those sheep were closely related to (i.e. targeted by) the methanogens in the vaccine. Based upon these findings, it was suggested that greater methane abatement might be possible if a greater proportion of the methanogen species/strains were targeted by the vaccine. Unfortunately, Williams *et al.* (2009) failed to significantly impact methane emission by these sheep, and the density of methanogens,

but reported a significant correlation between 16S rRNA gene sequence relatedness and cross-reactivity for the methanogens ($R^2 = 0.90$). Upon closer inspection, it would appear that the vaccine affected the diversity and composition of the methanogen population. This suggests that a highly specific vaccine can be made to target specific strains of methanogens, and that a more broad-spectrum approach is needed to be successful in the rumen. Williams *et al.* (2009) also reported that methanogens take longer than 4 weeks to adapt to dietary changes, which calls into question the validity of experimental results involving methanogens that were based upon 2–4-week acclimatization period normally observed for bacteria. Further studies are now warranted properly to assess the acclimatization period for the methanogenic archaea.

An additional vaccine has recently been developed using subcellular fractions of *Methanobrevibacter ruminantium* (Wedlock *et al.*, 2010). Twenty sheep were vaccinated and then re-vaccinated 3 weeks later and the antisera were found to cause agglutination of methanogens and decrease growth and methane production *in vitro*. To the best of our knowledge, *in vivo* testing of the efficacy of the newest vaccines against methanogens has not yet been conducted. Although this technology is high risk, it provides many options for long-term enteric methane abatement, as well as being a simple and cost-effective approach. Microorganisms which provide either hydrogen to methanogens (i.e. protozoa) or compete for hydrogen with methanogens (i.e. acetogens) are possible vaccine targets for the reduction of methane emissions.

Conclusions

Current mitigation strategies, in various stages of research, are focusing upon decreasing the hydrogen upon which methanogens are dependent, using alternative hydrogen sinks (Joblin, 1999), discovering anti-methanogenic compounds, eliminating the protozoa from the rumen and developing vaccine technologies against methanogens. While some of these approaches have been inconsistent or failed because of a lack of knowledge of the composition, function and microbial interactions within the ecosystem, other strategies, such as dietary lipids, have proved relatively successful.

The increasing demand for food from animal sources, particularly in the developing countries of the world, will lead to a continued high interest in the interaction of livestock with the environment. As a result, agriculture has the opportunity to help mitigate the challenge as well as adapt to the changes that will occur. Achieving meaningful reductions in methane emissions should be possible with advances in our knowledge of the intricacies of this complex ecosystem. New technologies are rapidly coming on line, which promise greatly to enhance the rate with which our knowledge of the ecosystem and its interrelationships will advance. High-throughput sequencing methodologies will greatly improve the rate of knowledge acquisition, which will enhance the ability to unravel species interrelationships. Novel options for reducing methane emissions, through control or elimination of methanogens, or other microorganisms, are likely to be developed through this improved understanding of the rumen microbiome.

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17 Emerging Contaminants in Livestock Manure: Hormones, Antibiotics and Antibiotic Resistance Genes

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Introduction

Most environmental concerns associated with livestock farms focus on nutrients (nitrogen, phosphorus) that impair the health of aquatic systems or on pathogens that may lead to food- or water-borne disease. In recent years, however, other potential contaminants are becoming of concern. Hormones (oestrogens, androgens, progesterone and various synthetic hormones) contained in livestock manure have generated wide interest because of their endocrine disrupting effects (Lange *et al.*, 2002; Hanselman *et al.*, 2003; Lorenzen *et al.*, 2004). Similarly, the extensive use of antibiotics in animal agriculture and the development of antibiotic-resistant bacteria are cause for increasing concern. Livestock operations are often cited as a reservoir for resistant bacteria and antibiotic resistance genes (Chee-Sanford *et al.*, 2001; Smith *et al.*, 2004; Sawant *et al.*, 2007; McKinney *et al.*, 2010), and antibiotic use has implications for both animal and human health. This chapter will focus on these emerging contaminants in livestock manure.

Hormones in Manure as Environmental Pollutants

Endogenous steroidal hormones

Hormones are synthesized in specialized glands of the endocrine system and are excreted at very low quantities in urine and faeces (Meyers *et al.*, 2001; Lintelmann *et al.*, 2003). The hormones in animal manure that have important environmental effects include oestrogens (oestrone, oestradiol and oestriol), androgens (testosterone) and progestagens (progesterone).

An estimated 49 t of oestrogens, 4.4 t of androgens and 279 t of gestagens were excreted by farm animals in the USA in 2002 (Lange *et al.*, 2002), and cattle production contributes about 90% of oestrogens and gestagens and 40% of androgens. About 1500 kg oestrone and oestradiol are excreted each year by farm animals in the UK (Johnson *et al.*, 2006), about four times more than the total oestrogens from humans. These estimates are uncertain, as data available may not be sufficient for accurate calculation of the total mass of oestrogens excreted (Hanselman *et al.*, 2003). It is clear, however,

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that large amounts of hormones from livestock farms are released into the environment each year (Table 17.1).

Exogenous hormonal growth promoters

In both humans and animals, sex steroids such as oestrogens, testosterone and progesterone regulate growth and development. This has led to the use of hormonal growth promoters (HGPs), natural sex steroids or their synthetic counterparts, in meat animals to increase feed efficiency and weight gain. Synthetic HGPs are widely used by the largest cattle-producing countries in the world, including the USA, Australia, Argentina and Canada (Silence, 2004). The first approved synthetic oestrogen was diethylstilbestrol (DES) in 1954. Because of its carcinogenic potential, DES was banned for all use in cattle production by the FDA in 1979 (Hayes, 2005).

Currently six different hormones are approved for such use in the USA, including three natural hormones (E2β, testosterone and progesterone) and three synthetic compounds that mimic the functions of these hormones (trenbolone acetate (TBA), zeranol and melengoestrol acetate (MGA); Le Guevel and Pakdel, 2001). The chronology of the use of these hormones in cattle in the USA is listed in Table 17.2 (Raun and Preston, 2002). The use of all of these is banned in the EU.

These HGPs are primarily used in the beef cattle industry, as exogenous androgens and oestrogens have little efficacy in pigs (Chaudhary and Price, 1987). In the dairy industry, progesterone-releasing implant devices are approved in lactating cows for oestrus synchronization (Price and Webb, 1988).

Naturally occurring HGPs

The direct effects of exogenous E2β administration on farm animals (calves, heifers, steers, lambs) include enhanced protein deposition in skeletal muscle and reduced nitrogen excretion (Meyer, 2001); growth performance is increased by 5–15%. Testosterone or other androgens are less active as compared to oestrogens in cattle or lambs, probably because there are fewer androgen receptors than oestrogen receptors (Sauerwein and Meyer, 1989). The anabolic mode of action of steroidal hormones is mediated directly by several organs and tissues (liver, bone, skin and other tissues) and indirectly via the somatotropic axis involving growth hormone and insulin-like growth factor I. The exogenous doses of these naturally occurring HGPs are less than the amounts naturally produced in mature bulls and pregnant cows, so their environmental risk is considered negligible (Avery and Avery, 2007).

Synthetic HGPs

Trenbolone acetate mimics the activity of testosterone and is administered as a subcutaneous

Table 17.1. Predicted total excretion of oestrogens from the human and farm animal populations in the UK (2004).

Type	Population (million)	Oestrone (kg year^{-1})	Oestradiol (kg year^{-1})	Discharge percentage[a]
Human	59	219	146	17
Dairy	2.2	693	365	49
Pig	5	367	19	18
Broiler chicken	112	15	34	2
Laying hens[b]	29.2	NC	NC	12
Breeding ewes	7.6	19	6	1.9
Non-breeding sheep	1.5	1.6	0.4	0.1
Total farm animals	157.5	1096	424	83
Total	216.5	1315	570	100

[a]Based on the total amount of oestrone and oestradiol.
[b]The combined amount of oestrone and oestradiol is 260 kg year^{-1}.
NC, not calculated because of insufficient data.

Table 17.2. Chronology of the use of anabolic agents in the US cattle industry.

Year	Event
1956	Oestradiol benzoate/progesterone implants approved for steers
1958	Oestradiol benzoate/testosterone propionate implants approved for beef heifers
1968	Oral melengoestrol acetate approved for beef heifers
1969	Zearnol implants (36 mg) approved for cattle
1982	Silastic oestradiol implant approved for cattle
1984	Oestradiol benzoate/progesterone implants approved for beef calves
1987	Trenbolone acetate implants approved for cattle
1991	Oestradiol/trenbolone acetate implants approved for steers
1993	Bovine somatotropin approved for use in lactating cows
1994	Oestradiol/trenbolone acetate implants approved for use in heifers
1995	72-mg zeranol implants approved for beef cattle
1996	Oestradiol/trenbolone acetate implants approved for stocker cattle

implant either alone or coupled with E2β. Zeranol, also called α-zearalanol, is a resorcylic acid lactone. Zeranol mimics the action of E2β and is often implanted as the sole compound. Melengoestrol acetate is administered as a feed additive, while all the other HGPs are administered as implants and can be used for oestrus synchronization and/or lactation induction in cattle as an active gestagen (Funston *et al.*, 2002). It is also fed to feedlot heifers to improve feed efficiency and weight gain (Kreikemeier and Mader, 2004).

Excretion of steroidal hormones

Oestrogens

Oestrogens are mainly excreted through urine and faeces. In faeces oestrogens mainly exist in free forms, while in urine oestrogens are mostly conjugated (Shore and Shemesh, 2003). Free and conjugated forms of oestrogen (17α-estradiol (E2α), E2β and oestrone (E1)) account for more than 90% of the excreted oestrogens in cattle (Hanselman *et al.*, 2003), but E2α is rarely excreted by swine and poultry. In cattle, 58% of the total oestrogen excretion is via the faeces (Ivie *et al.*, 1986), while swine and poultry excrete 96% and 69% of oestrogens in urine, respectively (Ainsworth *et al.*, 1962; Palme *et al.*, 1996).

There are several reports of the presence of steroidal hormones in animal manure suggesting contribution of the livestock industry to the environmental load of hormones. In early lactation cows, urinary excretion of total oestrogens was 174 μg day^{-1} (Erb *et al.*, 1977). In the 5 days prior to parturition, concentrations of E1, E2α and E2β in dairy faeces were 11.6, 60.0 and 33.6 ng g^{-1} dry sample (Hoffmann *et al.*, 1997). Three of four major oestrogens, E2α, E2β and E1 were detected in fresh faeces of both dairy and beef cattle (Wei *et al.*, 2011). Oestrone was the most prevalent in dairy faeces (101–865 μg kg^{-1} dry weight) while the concentration of E2α and E2β ranged from <1.1 to 1113 and from <1.9 to 485 μg kg^{-1} dry weight, respectively. Concentrations of these three hormones were lower in beef cattle faeces than in dairy faeces (<5.0–508, <1.1–260 and <1.9–243 μg kg^{-1} dry weight for E1, E2α and E2β, respectively). In cow manure, natural oestrogens were present at concentrations of 6, 17 and 16 ng g^{-1} dry solids for E2α, E2β and E1, respectively (Andaluri *et al.*, 2011).

Oestrogens are also detected in poultry manure with E2β being the most prevalent form followed by E2α and E1 (150, 93 and 44 ng g^{-1} dry solid, respectively; Andaluri *et al.*, 2011). Oestrogen content (oestradiol or oestradiol plus oestrone) of dry broiler litter of 28–30 ng g^{-1} has been reported (Casey *et al.*, 2004; Shore *et al.*, 1995). The concentration of E1, E2β and E2α in pig slurry was 243, 115 and 9 μg kg^{-1}, respectively (Laegdsmand *et al.*, 2009). Factors such as age, diet, season, health status and diurnal variation may contribute to variation

in excretion rates (Schwarzenberger *et al.*, 1996). Table 17.3 shows the calculated total daily excretion of oestrogens for cattle, pigs and sheep (Lange *et al.*, 2002).

Synthetic HGPs

As are endogenously produced steroidal hormones, exogenous natural or synthetic hormones administered to livestock are excreted in faeces and urine (Schiffer *et al.*, 2001; Wilson *et al.*, 2002). Zeranol and its metabolites appear in manure, with concentrations highest immediately after implantation (Baldwin *et al.*, 1983; Dixon *et al.*, 1986; Jansen *et al.*, 1986). Zeranol concentrations in manure declined steadily during the period 20–70 days, but were still detectable 120 days after implantation. About 10% and 45% of the implanted zeranol in steers was excreted through urine and faeces, respectively (Sharp and Dyer, 1972). Excretion of MGA is primarily through the faeces, but little data are available on persistence of excretion following cessation of administration. Tracer studies using radio-labelled MGA revealed that 60% of the dose is excreted (intact or as metabolites) in the faeces and 10% in the urine (Krzeminski *et al.*, 1981). For TBA, between 1.5 and 25% of the dose is excreted (average 8%; Schiffer *et al.*, 2001), mostly as the metabolite 17α-TBOH (Pottier *et al.*, 1981). This alpha form is the less biologically active of the two primary metabolites of TBA.

Degradation of steroidal hormones in manure during storage

The ultimate fate of manure nutrients is strongly influenced by how that manure is removed from the animal facility and how (and whether) it is stored; the same is true of steroidal hormones. Degradation of oestrogens is a complicated process and may include deconjugation, dissipation and mineralization (Ternes *et al.*, 1999; Lee *et al.*, 2003; Jacobsen *et al.*, 2005). Photodegradation has also been suggested as a mechanism of oestrogen dissipation and mineralization (Mitamura *et al.*, 2004; Feng *et al.*, 2005). Conventional manure handling systems are not designed to treat manure, but during prolonged storage the degradation of hormones can occur under anaerobic conditions.

Degradation in storage

The concentration of E1, E2α and E2β was 475, 98 and 104 ng l^{-1}, respectively, in liquid dairy manure stored in a lagoon for 8 months. The total amount of all oestrogens in the stored manure represented only 0.2% of estimated accumulation of oestrogens during 8 months of storage suggesting substantial degradation of oestrogens during storage in the lagoon (Zhao *et al.*, 2010). Oestrogen concentrations decreased significantly in broiler litter at pH 5 or 7 after 1 week (Shore *et al.*, 1995).

Yang *et al.* (2010) reported the degradation of steroid hormones (E2β, progesterone and testosterone) in swine manure by manure-borne bacteria. Oestradiol, progesterone and testosterone were completely degraded by 9, 17 and 21 h, respectively. In swine manure piles created over 5 days, the concentration of E1 was higher than that of E2β (29.4 versus 0.64 μg kg^{-1} dry weight) indicating the degradation or transformation of higher-degree oestrogens into E1; thus the oestrogen profile of manure piles or pits may not represent the actual proportion of oestrogens in fresh manure (Derby *et al.*, 2011).

The half-lives of the two metabolites of TBA (17α-TBOH and 17β-TBOH) were 267 days in liquid cattle manure (Schiffer *et al.*, 2001). Melangerol acetate was relatively stable in

Table 17.3. Estimation of total daily oestrogen excretion by farm animals (μg day^{-1}). Adapted from Lange *et al.* (2002).

Species	Category	Faecal excretion	Urinary excretion	Total excretion
Cattle	Cycling cows	200	99	299
	Bulls	360	180	540
Pigs	Cycling sows	14	100	114
Sheep	Cycling ewes	20	3	23

storage, with little degradation observed after 4.5 months of storage. Aeration accelerates hormone degradation. Testosterone degraded at a faster rate when swine manure was incubated aerobically than in anaerobic conditions (Yang *et al.*, 2010). Similarly, there are reports of reduced oestrogen concentrations in poultry and swine manure after aerobic composting (Hakk *et al.*, 2005; Derby *et al.*, 2011). The concentration of E1 in composted swine manure pile decreased and stabilized to <2 μg kg^{-1} dry weight after 36 days of storage, with a 94% reduction in E1 by 92 days. In contrast, E1 in un-composted manure decreased to a lesser extent (>5 μg kg^{-1} dry weight) and was not stabilized at 36 days. After 92 days storage, the E1 concentration was reduced by 78% in static manure. Total oestrogenicity was reduced by 79% and 74% in composted and static manure, respectively, indicating the advantage of storing manure before applying to the agricultural fields (Derby *et al.*, 2011).

Also, heat accelerates degradation. Zhao *et al.* (2010) reported that oestrogens were less frequently detected in streams during summer months and suggested rapid degradation of oestrogens in manure and soils at high temperatures. Total oestrogens were reduced by 80% in cattle faeces following 12 weeks of incubation at 20–23°C (Schlenker *et al.*, 1998).

Fate of hormones in manure following land application

When manure is land applied, some portion of excreted hormones may enter water systems through runoff or leaching (Nichols *et al.*, 1997; Finlay-Moore *et al.*, 2000; Dyer *et al.*, 2001; Jenkins *et al.*, 2006). All the hormones discussed above are at least partially degraded following land application of manure, to less active metabolites or all the way to CO_2. Their half-lives vary with the environmental matrix; half-lives of hormones in manure and soil are longer than in the aquatic environment. Temperature, moisture, oxygen and pH all influence degradation of steroidal hormones (Ying and Kookana, 2005).

Also important in evaluating risk to water resources is that steroidal hormones may be strongly sorbed to soil (Yu *et al.*, 2004) and sorption reduces the potential of leaching and runoff. Because manure is land-applied rather than discharged directly into waterways, the likely risk of hormones from livestock farms should be lower compared to hormones discharged from wastewater treatment plants. Oestrogens still have mobility to run off from soil, however, and much more research is needed on sorption of oestrogens and other steroid hormones to agricultural soils.

Dairy manure

The presence of 17β-estradiol (3300 ± 700 ng l^{-1}) was reported in runoff from plots amended with dairy manure (Dyer *et al.*, 2001). The concentration in the runoff reached 41 and 29 ng l^{-1} when manure was applied at nitrogen or phosphorus-based rate, respectively, 10–20-fold greater than in runoff from plots without manure application. Both free and conjugated oestrogens were detected in barnyard runoff on a dairy farm (Gadd *et al.*, 2010). Among major free oestrogens, E2α was the most prevalent form followed by E1 and E2β (110–11,000, 10–580 and 1–310 ng l^{-1}, respectively). Conjugated oestrogens were present in almost all the samples. The most prevalent oestrone-3-sulfate (12–180 ng l^{-1}) and less prevalent 17α-estradiol-3-sulfate and 17β-estradiol-disulfate (below detection limit to 230 and 17–320 ng l^{-1}, respectively) collectively represented 8% of the total oestrogen concentration.

Transport of oestrogen hormones through soil to groundwater is suggested by detection of E1 and E2β in the leachates from lysimeter soil columns after application of dairy farm effluents (Steiner *et al.*, 2010). The major transport mechanism was with dissolved or suspended organic matter (Laegdsmand *et al.*, 2009) through preferential/macropore flow pathways (Steiner *et al.*, 2010). Zhao *et al.* (2010) reported that most of the oestrogens in liquid manure are associated with the particles between 0.7 and 1.2 μm.

Poultry litter

One of the first reports that raised concern about hormones from animal agriculture was with land application of broiler litter (Nichols *et al.*, 1997). The 17β-estradiol concentrations in surface runoff increased with increasing application rate, reaching a maximum of 1280 ng l^{-1} at

an application rate of 7.05 t litter ha^{-1}. In another study (Finlay-Moore *et al.*, 2000), runoff concentrations of 17β-estradiol ranged between 20 and 2330 ng l^{-1} depending on broiler litter application rates and time between application and runoff, and soil concentrations of 17β-estradiol reached 675 ng kg^{-1}. The application of poultry litter did not substantially increase steroid hormone concentrations in soil but oestradiol and testosterone were present in subsurface drainage from the soil plots and runoff in ng l^{-1} concentrations (Jenkins *et al.*, 2009).

Both free and conjugated oestrogens were detected in surface runoff from soil receiving poultry litter (Dutta *et al.*, 2010) with the flow-weighted concentration of conjugated oestrogen (17β-estradiol-17-sulfate) higher (0.3–3.9 ng l^{-1}) than free oestrogens (0.75–2.5 and 0.5–1.9 ng l^{-1} oestrone and 17β-estradiol, respectively). The concentration of total oestrogens was lower in surface runoff when pelletized poultry litter was applied than when raw poultry litter was used (<1 versus 0.91–3.34 ng l^{-1}).

Swine manure

Natural oestrogens (E1, E2β and E2α) were detected in leachate from swine slurry-treated soil monoliths (vertical slice of soil in its natural position) with a range of <1 to 10 ng l^{-1} (Laegdsmand *et al.*, 2009). On average 0.0206% of total natural oestrogens applied to soil monoliths was leached after 16 week with faster leaching rate for the loamy soil than for the sandy soil.

Beef cattle manure – natural and synthetic steroidal hormones

Oestrogens, androgens and progesterones were detected in runoff following simulated rainfall on feedlots containing manure from beef cattle without exogenous HGP. The concentration of androgens and progesterone in the soil decreased by more than could be accounted for in runoff, suggesting rapid microbial transformation during rainfall (Mansell *et al.*, 2011).

Similarly to natural steroids, HGPs and their metabolites in manure sorb to soil, are subject to microbial and photo-degradation, but may also leach or runoff in certain conditions. Zeranol may be moderately sorbed to soil following manure application and it is partially mineralized; 50% of the applied zeranol was mineralized to CO_2 in 90 days after field application. Following land application of solid manure from heifers with TBA implants, parts per billion (ppb) concentrations of trenbolone were detected in soil samples up to 8 weeks after fertilization (Schiffer *et al.*, 2001). Following application of liquid manure, trenbelone metabolites were detected in soil at 8 days but not at 40 days. The increased persistency of TBA in solid manure was likely because its straw content provides more sites for sorption, limiting its degradation and possible leaching. Following application of manure from MGA-fed heifers, MGA persisted in soil for at least 6 months at ppb concentrations. Persistence in soil can be read two ways. On one hand, its persistence leads to questions of long-term environmental effect, but persistence also reflects strong binding to soil particles, reducing its risk of runoff and leaching.

Of greater concern than minimal losses from appropriately managed land application of manure would be runoff that occurs directly from the feedlot. Generally speaking, concentrations of HGP are found in ppb concentrations in faeces and in ppt concentrations in runoff from feedlots. Metabolites of TBA were detected in the discharge from feedlots in ppt (ng l^{-1}) concentrations (Durhan *et al.*, 2006). As in faeces, 17α-TBOH was more commonly detected (in six of nine sample times versus two of nine) and in higher concentrations (up to 120 versus 10–20 ng l^{-1}) than 17β-TBOH in the liquid discharge from a beef feedlot (Durhan *et al.*, 2006). Melengesterol acetate was detected in 6% of runoff samples from feedlots at concentrations ranging up to 115 ng l^{-1} (Bartelt-Hunt *et al.*, 2012).

Impact of steroidal hormones in streams and rivers

Endocrine disruptors in streams and rivers are of great interest and concern because of the direct exposure of wildlife and humans. Effects of these have been widely reviewed (Tabak *et al.*, 1981; Colborn and Clement, 1992; Colborn *et al.*, 1993; Hotchkiss *et al.*, 2008) and are summarized only briefly here. Concentrations of the steroid hormones discussed above were monitored in a national survey in 2002 (Table 17.4; Kolpin *et al.*, 2002).

Table 17.4. Steroid hormones reported in US rivers (Kolpin *et al.*, 2002).

Hormone	N	RL (μg l^{-1})	Freq (%)	Max (μg l^{-1})	Med (μg l^{-1})
17β-estradiol	85	0.5	10.6	0.2	0.16
17β-estradiol	70	0.005	5.7	0.074	0.03
Oestrone	70	0.005	7.1	0.112	0.027
Oestriol	70	0.005	21.4	0.051	0.019
Testosterone	70	0.005	2.8	0.214	0.116
Progesterone	70	0.005	4.3	0.199	0.11
cis-Androsterone	70	0.005	14.4	0.214	0.017

N, number of samples; RL, reporting level; Freq, frequency of detection; Max/Med, maximum/median detectable concentration.

Generally, hormones are detected in 3–20% of the river samples. Although the maximum concentrations are well below 1 mg l^{-1} for all hormones, this does not eliminate of the possible interactions of these hormones and other contaminants in the environment (Kolpin *et al.*, 2002).

Free oestrogens (total oestradiol and E1) and testosterone were detected (0.5–5 and 1–28 ng l^{-1}, respectively) in small streams draining from farm fields following application of poultry litter (Shore *et al.*, 1995). In a river adjacent to a feedlot, the concentration of oestrone, 17α-estradiol and 17β-estradiol was 900, 35 and 84 pg l^{-1}, respectively, 80 km downstream from the feedlot (Soto *et al.*, 2004). Both E2α and E2β were present in the streams receiving the drainage from 800-cow dairy farm (Zhao *et al.*, 2010). The presence of E2α and E2β was consistent throughout the spring months though at low concentration (<1 ng l^{-1}). Runoff due to melting of snow and thawing of frozen ground water during spring months might have moved oestrogens from soil to the streams. Oestrone (E1) and oestriol (E3) were not detected in streams (detection limit: 0.061 and 0.036 ng l^{-1}, respectively).

Zeranol was detected in rivers in Europe, where use of HGP is banned (Laganà *et al.*, 2004). Authors hypothesized that what they detected were in fact metabolites of the mycotoxin zearalenone. Following land application of manure from cattle treated with all three HGPs (TBA, MGA, zeranol), runoff from rainfall simulation contained residues only sporadically (less than 10% of the samples). When detected, these were in ppt (ng l^{-1}) concentrations in runoff (Biswas *et al.*, 2011). Exposure to steroidal hormones may alter or disrupt the functions of the endocrine system and cause adverse effects to organisms including mimicking or blocking receptor binding, or altering the rate of hormone synthesis or metabolism through interactions with the endocrine system (Meyers *et al.*, 2001). The effective levels of these hormones are as low as ng l^{-1} of water (Colborn *et al.*, 1993). For instance, a high incidence of intersexuality (feminization) was observed in a wide population of male roaches in the UK (Jobling *et al.*, 1998). Also, after exposure to cattle feedlot effluent, both testosterone synthesis and testis size decreased in male fathead minnows (Orlando *et al.*, 2004).

Groundwater contamination with steroidal hormones is rarely reported but can occur with leaching and runoff from fields or leaking from sewage systems (Jacobsen *et al.*, 2005). In sampling locations near a residential septic system, Swartz *et al.* (2006) detected 0.2–45 ng l^{-1} of 17β-estradiol and 0.4–120 ng l^{-1} of oestrone in the groundwater. The US Geological Survey (2005) reported a 60% frequency of detecting steroid hormones in groundwater from 47 sampling locations susceptible to contamination from either animal manure or human wastewaters. The concentrations of 17β-estradiol ranged from 13 to 80 ng l^{-1} in eight springs draining a karstic aquifer (Wicks *et al.*, 2004).

Best management practices to minimize loading to water

Ultimately, manure is applied to cropland or pastures as a fertilizer or soil amendment, with the goal of recycling manure nutrients through

crops. A variety of best management practices (BMPs) are used by farmers to reduce nutrient losses from their farms. Most states in the USA provide significant cost-share funds or tax credit programmes to support the implementation of BMPs. Their implementation is seen as a key component of strategies to reduce non-point-source nutrient pollution. Common BMPs include buffer strips, constructed wetlands and stream-bank fencing.

One recent report suggests that with a combination of BMPs implemented by a permitted concentrated dairy farm, surface waters downstream of the farm were no different from upstream waters in terms of oestrogenic activity (Shappell *et al.*, 2010). Maximum oestrogenic activity in samples from farm tile drains (≤ 0.257 ng l^{-1}) was twice as high as in the creek, but were only 25% of the proposed 'no observable effect' concentration for E_2. In many cases, the effects of these BMPs on the fate of EDCs are not known, but inferences can be drawn.

Constructed wetlands are marshes built to treat sewage, stormwater, leachate and increasingly, agricultural wastewater with the purpose of removing BOD, nutrients, metals and toxic organic compounds (Reddy and D'Angelo, 1997) and can be a potential means of removing steroidal hormones from agriculture wastewater. There are hardly any studies evaluating the effect of constructed wetlands on steroidal hormone removal but in a study with swine wastewater, oestrogenic activity (measured with the *in vitro* E-screen assay) was reduced by 83–93% with constructed wetlands and wetland outflow concentrations were below 3 ng l^{-1} (Shappell *et al.*, 2007).

Buffer strips are narrow strips of a permanent vegetative cover planted across a slope along a waterway to reduce runoff of nutrients, organic matter and pathogens from manure-amended fields to water bodies. Few data are available about hormone removal by buffer strips, but (Nichols *et al.*, 1998) observed reduced oestrogen mass loss from poultry-litter amended pastures after using buffer strips; reduction by 79%, 90% and 98% with buffer strips of 6-, 12- and 18-m width.

Stream bank fencing is the most common BMP to prevent the access of cattle to streams and other waterways reducing direct deposition of nutrients, pathogens and steroid hormones in manure. This practice can have significant negative economic impacts but researchers have found that simply providing an alternative water source will reduce the time cattle spend in streams by up to 90%, even without fencing (Sheffield *et al.*, 1997). While no data exist on the effect of preventing or limiting access to waterways on steroidal hormone deposition in streams, it is reasonable to expect a significant benefit.

One recent report suggests that with a combination of BMPs implemented, oestrogenic activity in surface waters downstream of a large permitted dairy farm were no different from in upstream waters (Shappell *et al.*, 2010). Maximum oestrogenic activity in samples from farm tile drains (≤ 0.257 ng l^{-1}) was twice as high as in the creek, but were only 25% of the proposed 'no observable effect' concentration for E_2. Because BMPs are so widely implemented and represent a significant investment of public funds, where effects on hormone fate are truly unknown more research is merited.

Antibiotic Use in Livestock and the Development of Antibiotic Resistance

Sub-therapeutic doses of antibiotics are fed to livestock for growth promotion and disease prevention and their use can reduce morbidity and mortality (Wileman *et al.*, 2009). However, the unintentional selection of bacteria that are resistant to powerful antibiotics could have important human health consequences, with several studies noting identical resistance elements in both humans and food animals (Boerlin *et al.*, 2001; van den Bogaard *et al.*, 2001; Lauderdale *et al.*, 2002; Ho *et al.*, 2010). The use and environmental impact of antibiotic use in food animals has been extensively reviewed (Sarmah *et al.*, 2006). Some of the more common uses of antibiotics and their environmental impact are summarized here with a focus on sub-therapeutic and growth promotion uses.

The inclusion of oxytetracycline (a tetracycline) and neomycin (an aminoglycoside) in milk replacer for young calves is one of the most common uses of sub-therapeutic antibiotics in the US and Canadian dairy industries. About 70% of calves in the USA are fed milk replacer (USDA, 2007) and 80% of milk replacers used contained antibiotics. Other tetracylines are used widely for prevention of disease in poultry, swine and

sheep, though their use is banned in the EU and Australia. The macrolide tylosin is used in swine, beef cattle and poultry.

Ionophores (e.g. monensin, lasalocid) are antibiotics commonly fed to beef and dairy cattle to increase weight gain and feed efficiency, and to various species of poultry for control of coccidiosis (Table 17.5). They have been used in growing dairy and beef cattle since 1977, and in lactating dairy cows in Canada since 1997 and the USA since 2004. They are used in virtually all beef feedlots in the USA and fed to 58% of dairy heifers (USDA, 2007). Their use is not allowed in the EU and Australia.

Most antibiotics are poorly absorbed from the gastrointestinal tract so are largely excreted via faeces (Sarmah *et al.*, 2006). Chlortetracycline and tetracycline were detected in ppm (mg l^{-1}) concentrations in swine manure leaving the flush barn and in the holding pond (Zilles *et al.*, 2005).

Potential for runoff of antibiotics following land application of manure

The potential environmental impact of various excreted antibiotics is affected by their relative degradability and ability to sorb to manure and soil particles (Sarmah *et al.*, 2006). Many land-applied antibiotics are polar compounds but not highly soluble in water, so they become highly sorbed to soil. For instance, tetracycline, oxytetracycline and tylosin bind tightly to soil and are relatively stable if land-applied (Kemper, 2008). Soil concentrations are, therefore, usually higher than in water, ranging from ppb (μg kg^{-1}) concentrations in soil for macrolides and fluoroquinolones to nearly ppm (mg kg^{-1}) for tetracyclines (Kemper, 2008). While antibiotics bound to soil are not prone to leaching and their runoff can be controlled if erosion is controlled, soil does act as a reservoir of pollutant that may migrate to the aqueous phase over time. For antibiotics that sorb tightly to soil, BMPs such as buffer strips and sedimentation traps may be especially effective in controlling loss (Smith *et al.*, 1994; Chu *et al.*, 2010).

Table 17.5. Approved ionophores for use in the USA. Adapted from McGuffey *et al.* (2001).

Ionophore	Species approved
Laidlomycin	Cattle
Lasalocid	Poultry, cattle, rabbits, sheep
Monensin	Poultry, cattle, goats, lactating dairy cows
Maduramicin	Poultry
Narasin	Poultry
Salinomycin	Poultry
Semduramicin	Poultry

Less tightly sorbed to soils are the ionophores, especially monensin (Lee *et al.*, 2007). Sulfonamides stand out for their relatively low sorption potential (Sarmah *et al.*, 2006), and their soil concentrations are relatively low (in the low ppb in soil). These are considered relatively mobile in soil.

Degradation of antibiotics in manure and soil is largely microbial (Sarmah *et al.*, 2006) and is influenced by temperature, rainfall and soil properties. As with steroidal hormones, degradation is faster under aerobic conditions than anaerobic (Lee *et al.*, 2007). The half-lives of tetracyclines are in the order of weeks, while monensin has a half-life of ~2 months in anaerobically stored manure. Once land applied, however, monensin is rapidly degraded with a half-life of 2–3 days (Lee *et al.*, 2007). Tylosin is rapidly degraded with a half-life in manure and manure soil mixtures of days to weeks (Zilles *et al.*, 2005; Sarmah *et al.*, 2006).

Rainfall simulation studies with antibiotics directly sprayed on soil suggest that the transport of different antibiotics in soil and to runoff varies significantly (Kim *et al.*, 2010). Monensin was susceptible to runoff in the aqueous phase, while tylosin and erythromycin were most susceptible to runoff bound to sediment. (This report is not contradictory to the common observation that tylosin is rapidly degraded in manure, because the runoff was collected following direct application of antibiotic to plots.) No sediment-bound sulfathiazole was found in runoff, and tetracycline and chlortetracycline were found only in low concentrations in sediment-bound samples taken at 5, 10 and 15 min into the rainfall event. Instead, these three compounds were more likely to be transported to subsurface soil. Varying behaviour of different antibiotics in soil will require different control practices to prevent environmental contamination.

Antibiotics have been detected in ground and surface water, usually in ppb concentrations (Sarmah *et al.*, 2006). In a sampling survey of rivers in 23 watersheds in Alberta, Canada, parts per trillion concentrations of ten antibiotics were detected in a total of 51% of samples (Forrest *et al.*, 2011). Monensin was the most frequently detected (34% of samples) at concentrations to 853 ppt. These concentrations are orders of magnitude lower than concentrations considered ecotoxic, but data in this area are scarce and indirect or cumulative effects on an ecosystem are possible.

Development of antibiotic resistance in gut and faecal bacteria

Increasing antimicrobial resistance in manure of mature cattle (Alexander *et al.*, 2009) and other livestock (Marshall and Levy, 2011) has been observed with feed antibiotics. In calves fed 'waste milk' containing penicillin residues, resistance of faecal bacteria to penicillin increased with increasing dose fed (Langford *et al.*, 2003). Berge *et al.* (2006) found that milk replacer containing neomycin sulfate and tetracycline selected for *Escherichia coli* resistant to classes of antimicrobials not used, including aminoglycosides, chloramphenicol and sulfonamides.

In the Netherlands, resistance of *E. coli* to antibiotics used regularly in poultry in the faeces of broiler, turkey and layer indicated that resistance was more prevalent in broiler and turkey populations given antibiotics routinely than in layer populations where antibiotics are rarely used (van den Bogaard *et al.*, 2001). Resistance to nearly all antibiotics tested was higher in faecal *E. coli* of turkey and broiler farmers than in laying-hen farmers, and was also elevated in people who slaughter turkey and broilers. Genetic similarity between the resistant *E. coli* in faeces of broilers, turkeys and the people who work with them supported the hypothesis that resistant clones and resistance plasmids of *E. coli* may be transmitted from poultry to humans.

In contrast, a recent culture-independent (qPCR) evaluation of the effect of antibiotic addition in milk replacer on antibiotic resistance in calf faeces through weaning showed little difference with antibiotic treatment (Thames *et al.*, 2012). Faeces of all calves carried at least one gene encoding resistance to each of the three classes of antibiotics of interest (tetracylines, sulfonamides, erythromycins), and faecal samples from all calves carried multiple types of antibiotic resistant genes (ARGs) within the same sample. Among the eight ARGs examined across three classes, *tet*O was the only one to exhibit an increase in response to antibiotic treatment, and only when normalized to 16S rRNA genes. These results suggest that removal of chlortetracycline and neomycin from milk replacer will not eliminate shedding of ARGs into the environment. Similarly, McKinney *et al.* (2010) observed only modest reduction of *tet* and *sul* ARG abundance in waste lagoons on organic dairy farms relative to those of conventional dairies.

While bacteria can adapt to ionophores (Houlihan and Russell, 2003), genes coding for resistance to any ionophore have not been identified. Also, the use of ionophores in cattle feed does not necessarily lead to other types of antibiotic resistance (Houlihan and Russell, 2003).

Legal restrictions on feeding antibiotics

In the USA, the Food and Drug Administration (FDA) recently issued new rules limiting antibiotic use in milk replacers for calves (21CFR § 520.1484; 21CFR § 520.1660d). The impact of these changes is not yet clear, but the complete ban on sub-therapeutic use of antibiotics in the EU in the late 1990s provides a case study of long term effects (Boerlin *et al.*, 2001). A major trigger for earlier bans in Denmark and other Nordic countries was linkage of avoparcin use in broiler chickens and swine with deadly vancomycin-resistant *Enterococci* infection in humans (Bager *et al.*, 1997; Bates, 1997). In the 3 years following the European ban, a slight decrease in antibiotic resistance was observed in rectal swabs collected from food animals at slaughter, and a documentable decrease in acquired faecal enterococci resistance in humans was observed (Casewell *et al.*, 2003). Following the Taiwan avoparcin ban in chickens in 2000, the incidence of VRE decreased significantly (Lauderdale *et al.*, 2002). However, VRE still persisted following the ban and resistance to other classes of antibiotics, including tetracyclines and macrolides, stayed the same or even increased. In support of this observation, vancomycin and macrolide ARGs have been noted to be present

together on the same genetic element from isolates obtained from Danish pig herds (Aarestrup *et al.*, 2000). Thus, it is critical to consider co-selection of antibiotic resistance across antibiotic classes in evaluating impacts of feed antibiotics.

Sub-therapeutic antibiotics and animal health

Effects of sub-therapeutic antibiotic use on animal health have been difficult to assess and the general consequences of antibiotic amendment in livestock feed are complex (Marshall and Levy, 2011). Based on medicated feed prescription rates, decreased rates of respiratory infections were observed in Swiss piglets and fattening pigs in the 3 years following the antibiotic ban relative to the 3 years prior, but gastrointestinal tract infections appeared to go up (Arnold *et al.*, 2004). A marked increase in necrotic enteritis was noted in broiler chickens in Norway following the avoparcin ban, but this seemed to be ameliorated by the subsequent approval of narisin as a feed additive (Grave *et al.*, 2004). Also, while therapeutic antibiotic use rates did indeed decrease in Switzerland (Arnold *et al.*, 2004), Norway and Sweden (Grave *et al.*, 2004) following the EU ban, therapeutic use increased in weaning piglets in Denmark (Grave *et al.*, 2004). Generally, a comprehensive examination of banning sub-therapeutic antibiotic use in piglets, beef cattle and poultry in Sweden indicated that although temporary health problems and increases in antibiotic uses were noted, strategic management practices make possible competitive animal production and reduced antibiotic use (Wierup, 2001).

The effect of milk replacer medication on calf growth and health depends partly on the nutrient density and intake of milk replacer. Calves fed nutrient-dense ‘accelerated’ milk replacers did not benefit from antibiotic inclusion (Thames *et al.*, 2012), while those fed traditional milk replacers usually show improved growth and health (Quigley and Drew, 2000; Berge *et al.*, 2006; Stanton *et al.*, 2010). Also, quality and quantity of colostrum feeding influences the benefit of antibiotic addition (Kaneene *et al.*, 2008). In situations where management or nutrition is not optimal, health benefits with milk replacer medication may be more apparent.

The FDA approved the ionophore monensin for use in lactating cows in 2004, partly because of improvements in cow health. Monensin has been shown to reduce the incidence of clinical ketosis, reduce the duration of subclinical ketosis, reduce the rate of new mastitis infections and reduce the incidence of displaced abomasums (McGuffey *et al.*, 2001). Ionophores also contribute to control of protozoa that cause coccidiosis in poultry and calves.

Conclusions

The environmental effects associated with hormones and antibiotics from livestock farms have induced great interest and concerns in scientists, governments and the general public. Oestrogens, androgens and progestagens have been detected in some US rivers (Kolpin *et al.*, 2002) and may cause endocrine disrupting effects, especially in aquatic species. Antibiotics have also been detected, in ppb or ppt concentrations in ground and surface water. Although degradation of both hormones and antibiotics occurs during manure storage and after land application, more research is needed on effective approaches to reduce loading of these from livestock farms. Also, more work is needed on the environmental fate, potency and biological effects of these compounds originating from livestock farms.

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18 Animal Agriculture: How Can It Be Sustainable in the Future?

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> When nature works on her own, she only creates living soils. But, the moment the human being enters, they can either work like nature and rebuild the life of the soil with every action, or take it away. The choice is desertification or fertile and living soils that feed us.
>
> Dr Vandana Shiva, physicist, environmental activist and anti-globalization author in the documentary film, 'Dirt! The Movie' (2009)

Introduction

Animal agriculture as currently practised and managed in many areas of the world is not sustainable in the future. To make animal agriculture more universally sustainable it is imperative that the close proximity, interrelationships, cycles and balances among soil, water, plants and animals be reemphasized, revitalized and developed to build and maintain healthy soils and a continuum of holistic sustainable animal agricultural systems (also known as 'mixed plant–animal systems') in developing and developed countries.

Several variations on the definition (concept) of sustainable agriculture and sustainable animal agriculture are offered in this book. Virtually all conceptually encompass the qualities of manageable, viable and equitable interrelationships among the three primary pillars of sustainability – social, environmental and economic – people, planet and profit (Chapter 1). Most accurately, sustainable agriculture and assuredly sustainable animal agriculture belong under the umbrella of sustainable development (Raman, 2006). The Bruntland Commission Report – *Our Common Future* (WCED, 1987) provided the overarching definition of sustainable development as 'development that meets the needs of the present without compromising the ability of future generations to meet their own needs'. Furthermore, it was stressed that sustainable development is not possible without sustainable agriculture (Raman, 2006). Sustainable animal agriculture is a crucial component of sustainable agriculture because animals are an irreplaceable component in the interplay among mass, including especially water and other nutrients, and energy recycling in mixed plant–animal systems. Agriculture and

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agricultural sustainability are indispensable to achieve sustainable development (social sustainability) regionally, nationally and globally.

An excellent operational framework for sustainable agriculture in both developed and developing countries is embodied in the four primary imperatives set forth by the US National Research Council's expert working committee in the report *Toward Sustainable Agricultural Systems in the 21st Century* (NRC, 2010). In my view, these imperatives must be adopted and actively pursued in animal agriculture in the next 50 years in both developing and developed countries for animals to be part of the sustainable agriculture. These imperatives are: (i) provide a secure, safe supply of food, feed and fibre for humans and contribute to energy needs (e.g. biofuels) (people); (ii) enhance the quality of the physical environment (land, water and air) and natural resource base (planet); (iii) foster and support the economic viability and vitality of agriculture and farmers (profit); and (iv) enhance the quality of life of farmers, farm workers, and agricultural, rural and urban communities, and whole societies (people).

The objectives of this concluding chapter are: (i) to revisit with readers the long-held basic conceptual framework for sustainable agriculture and nutrient cycling; (ii) to address whether animal agriculture can have a fundamental and essential role in that framework in the future; (iii) to emphasize particularly the essentiality of one key natural resource – water for sustainability – and to raise the question as to whether animal agriculture must change significantly its water use if it is to be a significant component of a future sustainable agriculture; and (iv) to encourage all of us to 'go back to basics' by placing animal agriculture in the context of a continuum of different local holistic, sustainable mixed plant–animal systems that effectively recycle nutrients and energy, as influenced by local setting and available and implementable technologies, however simple or sophisticated. The chapter concludes by encouraging students, scientists, animal farmers, all agriculturalists, communities and societies and thus policy makers, in both developing and more developed countries to embrace the beginning of the formative transformation and continuation to sustainable animal agriculture for the mid-21st century.

Back to Basics

Schools of thinking about agriculture sustainability

Three decades ago, Douglass (1984), in the volume *Agricultural Sustainability in a Changing World Order*, wrote about three specific 'schools of thinking', which provide the foundation for a working definition of sustainable animal agriculture into the future. Each school represented a quite different viewpoint or belief about sustainability:

1. The notion of the first school is that sustainability means food security with continuously improving agricultural productivity and efficiency to meet ever-increasing demands for food for a growing population; with the determinism that 'good' food security outcomes rationalize (justify) other consequences – negative or degenerative – that might occur (e.g. water and air pollution, loss of topsoil, etc.).
2. The second belief focuses on the absolute necessity of stewardship of the environment (maintaining the natural ecology) before agriculture can possibly be or become sustainable.
3. The third conviction espoused by 'alternative' agriculturists is that preservation of natural resources, promoting rural cultures and fostering self-reliance are utmost – all of which they argue are intrinsic to preservation of family farming.

In the introduction to the book, Douglass (1984) then proposed a conceptual compromise or integration among these schools (Fig. 18.1): 'That agriculture will be found to be sustainable when ways are discovered to meet future demands for foodstuffs without imposing on society real increases in social costs of production and without causing the distribution of opportunities and income to worsen.' Measured against these standards and based on the evidence available at the time, he concluded that then-existing agriculture production systems were not sustainable (Douglass, 1984). How would we answer 30 years later?

The first school of thinking embodies sufficiency of food security through plant and animal production in the farm, opportunities to manage and improve productivity, and

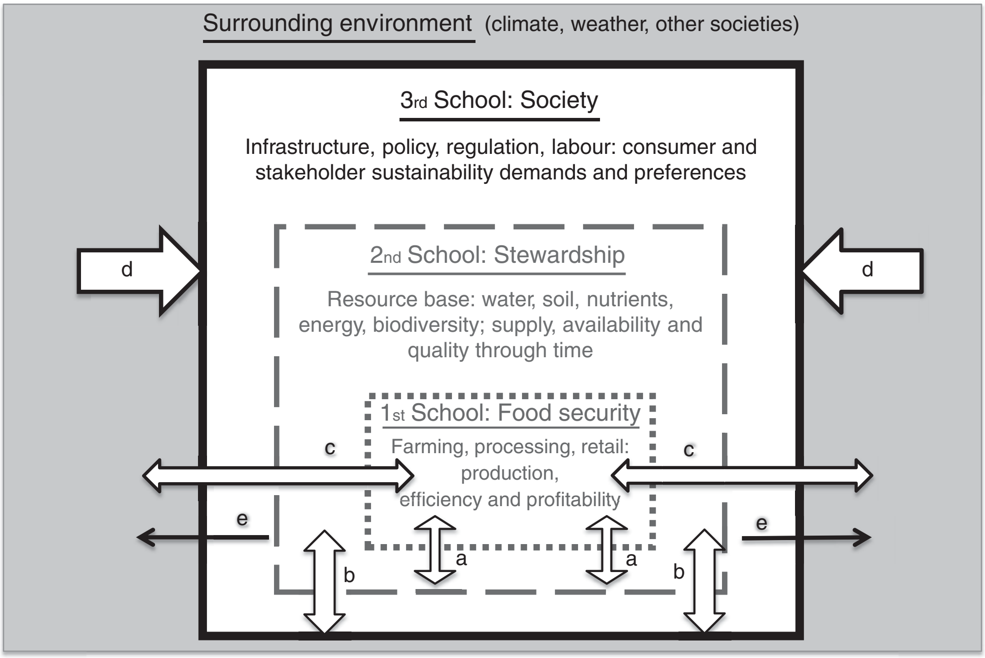

Fig. 18.1. Illustration of the conceptual framework of schools of thinking and their interactions in a holistic sustainable agroecosystem. Adopted from Hollmann (2010) and writings of Douglass (1984) and Lowrance *et al*. (1986). Arrows indicate interactions among schools of thinking. Note that conceptually the surrounding environment may affect the lower level schools or systems in major ways (e.g. arrows 'd'); whereas, the lower levels may have relatively little influence on the surrounding environment, unless aggregated and primarily in uncontrollable ways (arrows 'e'). Emerging schools combine lower schools (now subsystems) with parts of their surrounding environment. As a consequence, subsystems interact and influence one another actively and directly (arrows 'a', 'b' and 'c').

conversion of inputs to outputs, following through to processing and retail with the ultimate driver being profitability at various stages of the supply chain. As such, agriculture is obligated to produce sufficient amounts of food acceptable for humans and feed for livestock, while providing sufficient income for farm operators and workers, processors and retailers. This first school of thought captures a 'sustainability' based primarily on the market drivers of supply and demand, on profitability and on technological advancements needed to ensure ever-rising yields and marginal improvements in outputs over inputs (improved efficiency). Followers or proponents of this school believe that conventional, commercial agriculture and its market systems unequivocally will foster sustainability. They may be sceptical of new or different needs, ideas or practices outside the norms of current mainstream agriculture (Thompson, 2007); especially if potential changes or advances might increase costs of production or otherwise affect short- or intermediate-term markets or profitability. Implicit in the first school is the assumption of an unlimited resource base (e.g. water, soil, nutrients, energy, biodiversity), a very robust if not indestructible environment; and, perhaps if some deterioration of the environment does occur, it is a necessary sacrifice for the larger causes or objectives of food security and profitability.

The second belief maintains that there is a fixed (finite) supply, availability and quality of the resource base through time; and that resource depletion and (or) environmental damage are risks, but they are not acceptable. In this school, sustainability is regarded as stewardship (Brown, 1984). Whereas the first school focuses on food security and profitability by maximization of

output per input, the stewardship belief adds the relevance of a continuous temporal component and views sustainable agriculture as optimal management of the resource base (Berkes and Folke, 1998). According to the stewardship school, agricultural production has an environmental cost, and neither the resource base nor the environment can be (must be) depleted (or even damaged) to achieve food security and profitability for agriculture. Management must strive to optimize yield (output) and efficiency of resource use through an infinite period, while keeping the environment intact, or even enhancing it.

The third school of thinking incorporates communities and societies (the public, policy makers, stakeholders and consumers) with the requisite expectations of what sustainable agriculture and food production must be. In this belief, agriculture absolutely is not the primary entity unto itself, but rather embedded in a larger system with other (sub-) systems, all relying on the same limited resource base (second school). The third school is the most integrated with interdependent components; sustainable agriculture is viewed as wholly interactive with society and vice versa – members of society as stakeholders have varying degrees of active, more passive, or reflexive (e.g. purchasing decisions) participation, expectations and perceptions about sustainable agriculture and agriculture production (Fig. 18.1). At minimum, society fulfils the role of providing infrastructure (e.g. transportation and communication networks, government support and financial institutions, and associated industrial service-providers, and consumers of agriculture products) and governmental assurances (e.g. policy, enforcement of laws and regulations, monetary, banking and insurance systems). In reciprocation, farmers provide not only food security for people in return for income, but also support local communities in participant and leadership roles, and with jobs and public services. Sustainable (animal) agriculture in the context of the larger system (highest level of hierarchy or integration) has a requisite position in sustainable development of rural and rural–urban communities, and societies.

Embrace and adopt systems theory

Under the assumption that there is an ethical obligation to assure sufficient food for human beings now and in the future, agriculture must produce necessary amounts of food while minimizing degradation of natural and human environments. The emergence of the second school (stewardship) from the first (food security) in Douglass's (1984) discussion illustrates the general approach of systems theory – the merging of some number of lower level (sub-) systems leading to aggregation into a broader (larger) holistic system. Food production (first school) is strictly in the (sub-) system, surrounded or encompassed by the natural physical environment that provides energy, air, water, fertile soil, nutrients and biodiversity (Fig. 18.1). However, food production actively interacts with these resources in the environment and alters the inventory, supply, availability and quality of the resource base; for example, under- or over-application of animal manure may affect soil health, surface and ground-water quality, and biodiversity. Therefore, to develop and practise holistic systems thinking the farm (sub-) system (in the first school) must absolutely include the resource base (in the second school) as a (sub-) system of the larger aggregate system of society (the third school) all surrounded by the environment (Fig. 18.1). This ensures that it is well understood that achieving food security through food production does in fact alter the resource base and the natural environment. Conversely, narrowing the thinking about food production to just production efficiency (output/input) within the surrounding environment that provides the resources neglects the complexity of the dynamics and interactions between food production and the resource base. This highlights the necessity of assessing food production in animal agriculture, from a holistic perspective, more than simply just from the perspective of animal or farm (sub-) system productivity and efficiency. Lowrance *et al*. (1986) working from Douglass (1984) wrote of these as a more hierarchical construct for sustainable agriculture: sustainability as food sufficiency (lowest level of aggregation); sustainability as stewardship; and sustainability as community to accommodate competing definitions and embrace holistic thinking.

Fundamentals of nutrient and energy cycles

In the broad context, production and utilization of agricultural products (food, feed and fibre) from plants and animals is an extension and the continuation of the managed flow of nutrients

and energy (e.g. organic residues) from nature. The concept of cycling of nutrients and energy within a farm is fundamental and vital to its potential to be a sustainable (sub-) system. Sustainable agricultural production must be viewed as a managed flow of energy and nutrients among soil, plants and animals within a cycle in a whole farming system. Flow is cyclic, although at times, it may not be continuous and sometimes is incomplete, or there may be some leakage. Management of the inevitable leakage from the cycle via soil, water and air is crucial to the (sub-) system's degree of potential sustainability. Flow may be strictly within one farm (sub-) system or it may be across levels of aggregation as illustrated and described by the interaction arrows 'a', 'b' and 'c' in Fig. 18.1.

Figure 18.2 is the illustration of an example mixed crop–animal system (dairy farm) used as an extension education tool in Michigan (Michigan State University Extension Dairy Team https://www.msu.edu/user/mdr/nutrientcycle.html). In particular, the cycle illustrates the major qualitative flows of phosphorus (P, and other nutrients) and four key management critical control points in the most common dairy farm system currently in Michigan. The cycling and recycling are by no means continuous or quantitatively complete because there are inflows, outflows and movement across the farm boundary, and there is risk and may be leakage from the (sub-) system. Sources of energy and nutrients (e.g. purchased feeds and fertilizers) are transported into the (sub-) system; and outflows such as milk and animals are exported across the farm boundary. Additionally, other nutrients such as nitrogen (N) leave the farm either by intentional transport such as in milk, animals or manure, or may be lost to air and surface and ground-waters with potential risk of pollution. Inevitably, a significant fraction of N from manure volatilizes into air; whereas, other

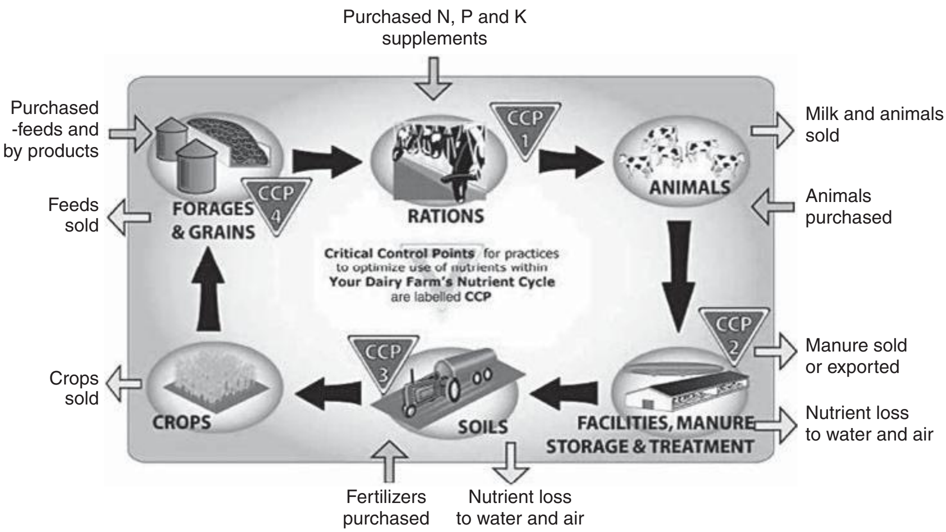

Fig. 18.2. Simplified example of a farm nutrient cycle of a mixed crop–animal (dairy) sub-system. Courtesy of Michigan State University Extension Dairy Team (2006) (https://www.msu.edu/user/mdr/nutrientcycle.html). The shaded area presents a single farm (sub-system) within its boundary. Large dark bold arrows indicate the flow of nutrients and energy in a cycle. Smaller arrows show the import and export flow of products (nutrients and energy) back and forth across the farm boundary. Four critical control points (CCPs) denote farm practices to optimize or maximize management of and use of nutrients as: CCP1, use accurate and precise ration formulation and feeding management; CCP2, capture and retain nutrients in the farmstead to eliminate or reduce nutrient pollution into water and air; CCP3, maximize nutrient uptake in the plant root zone by strategic and timely application of manure and commercial fertilizer; and CCP4, minimize loss of nutrients and energy during harvest and storage of crops whether for animal feed or for export sale.

N may be reintroduced or introduced into the cycle by fertilization (e.g. from manure or commercial synthetic fertilizer), by N-fixing plants, or by deposition from the atmosphere.

In the past (e.g. in the USA – late 1800s–1940s), the cyclic flow of nutrients more or less occurred typically in individual multipurpose farms without much notice or documentation. Virtually all of these farms were by necessity mixed crop–animal systems. After the Second World War, industrialization of agriculture began to varying degrees in different regions. This began to change the mixed plant–animal system paradigm. Whereas many of these so-called 'industrial farms' could excel at increasing productivity and efficiency and were increasingly significant contributors to the food supply at relatively inexpensive food prices, nutrients (e.g. feed and fertilizer nutrients) travelled to these farms from farther away distances, and nutrient cycles occupied larger land areas as some farms became more specialized and concentrated their human and financial capital in more specialized and intensive farming operations. Many farms concentrated efforts and investments almost entirely in either crop farming or animal farming. This necessitated a need for feed crops, ration supplements (e.g. speciality ingredients with denser concentrations of high-quality protein, minerals and vitamins). Consequently, nutrients were transported long distances (at least partially because of relatively inexpensive, government-subsidized fossil fuels) to these specialized animal farms (Naylor *et al.*, 2005). This resulted in the greater concentration of nutrients and animals in smaller, finite land spaces. Eventually, some areas predominated for crop production, whereas others focused more on animal production. With this shift the once cyclic flow of nutrients among the soil, and crop and animal enterprises within some single farm (sub-) systems was at risk of becoming disconnected or even truncated (Naylor *et al.*, 2005).

As a consequence, nutrients (e.g. P, N, K) moved from crop-producing areas to some animal-producing areas, but were not effectively recycled within the (sub-) system or moved back to their origin; thus, in some cases they accumulated over time, and water and air pollution problems arose (Chapters 7, 8, 9, 10 and 11). Consequently, these nutrients were accumulating in regions where animal agriculture was predominant without enough space or plants to complete the cycles for healthy soil building to fortify soil tilth. Also, some crop-producing areas became depleted of needed nutrients, or appreciable application of synthetic fertilizers replaced the nutrients from animal manure that previously were recycled to cropland.

Now federal and state regulations (e.g. US Clean Water and Clean Air Acts), largely consequences of breaking nutrient cycles in individual farm (sub-) systems or aggregate farm systems, have been implemented to hold farms (especially large farms) accountable for effective management of manure nutrients and to avoid over-application and leakage of nutrients (e.g. Chapters 7, 8, 9 and 10). Compliance with regulation increased and continues to increase, making export of manure nutrients necessary when local land base is insufficient for proper application of nutrients, installation of new technologies to deal with excess nutrients, implementation of mitigation techniques, or in some cases depopulation from livestock farms in nutrient-dense regions. All of these additional management practices impose exorbitant costs in the short- and medium-terms on farmers, agriculture in general, rural communities in the affected regions, and ultimately on society. Much of this could have been and can be avoided if the soil, plants and animals were kept in relatively close and economic proximity. Many of these additional costly practices are relatively temporary incremental short-gap fixes – export of nutrients, mitigation techniques and depopulation – that must give way to a much larger transformation in the future (see subsequently).

Potential Benefits of Animal Agriculture

Animal agriculture in very extensive to very intensive production systems potentially has at least three significant attributes beneficial to humankind. First, and most obviously, animals provide food, fibre, livelihoods, prosperity and enjoyment for humankind. It also is known that as wealth and standard of living increase in many societies people's diets are comprised of larger proportions of food derived directly from animals (FAO, 2005; Steinfeld *et al.*, 2006; McMichael *et al.*, 2007; Garnett, 2009, 2010).

Second, and often overlooked or poorly understood by some, animals have a critical role in natural and man-managed nutrient and energy (e.g. organic residues) cycles combined with and in synergy with the sun, water, air, plants and the landscape to foster and enrich soil health and productivity (Janzen, 2011). In recent years, the essential nature of having animals as part of whole systems (natural or man-managed) has been increasingly addressed and demonstrated (Savory Institute, 2012). This is being examined with increasing and thorough scientific scrutiny. The physical, fertility (nutritional) and energetic influences of animals in sustainable agro-ecosystems are coming to the forefront. Last, especially crucial to a future sustainable animal agriculture, are the ruminants (e.g. cattle, goats and sheep, etc.). The microbial fermentation in their forestomachs (rumen and reticulum) breaks down structural carbohydrates and can use low-quality protein (e.g. non-protein N sources) to yield microbial protein and energy substrates (e.g. volatile fatty acids) absorbed and used by the animal's tissues for maintenance, growth, lactation and pregnancy. Ruminants also are able to utilize biomass grown on land not suitable for effective direct production of human-edible foods or biofuels and convert it to high-quality animal proteins with associated dietary essential fats, vitamins and minerals, and energy for humans.

Trends in animal production and societies

Animal agriculture has both positive and negative influences on the natural resource base, human health, social equity and economic growth (World Bank, 2009). It is one of the most rapidly growing sectors in developing countries, already a third of the total gross domestic agriculture production, and increasing. Also, global animal agriculture utilizes about a third of the earth's land area and has an estimated asset value of over US$1.4 trillion (Steinfeld *et al.*, 2006). However, there is a vivid dichotomy currently in the characteristics of animal agriculture between developed and developing countries. In more developed countries globally connected market-value chains have emerged in recent decades (Chapters 1 and 12), whereas, in less developed countries, animals directly support and enhance the security and standard of living of an estimated 600 million small farmers predominantly on a local basis (Thornton *et al.*, 2006). Total meat production in developing countries tripled between 1980 and 2002; for example, in eastern Asia with rapid growth of poultry and swine production (World Bank, 2009). However, in developed countries production and consumption of animal products are increasing only very slowly or have plateaued, although still available in significant quantities. In more industrialized countries slightly over half of the agriculture gross domestic product is from animal agriculture and associated upstream merchandizing (World Bank, 2009).

Demand

It is well established that primary factors increasing future global demand for animal products are: (i) population growth; (ii) urbanization; and (iii) increasing disposable incomes (Delgado, 2005; Steinfeld *et al.*, 2006). Global population is projected at more than 9 billion people by 2050 compared with about 6.8 billion in 2010 (United Nations Population Division, 2012). Second, the vast majority of growth (over 70% increase) is to occur in urban areas with simultaneously declining (about 13%) rural populations. Urban growth is forecasted to be greatest (over 80% increase) in Asia (e.g. China and India) and sub-Saharan Africa (greater than twofold increase). Urbanization in developing countries with associated increased infrastructure for transportation and food preservation will allow greater distribution of perishable animal products (Delgado, 2005). In contrast, human population numbers in Latin America and the Caribbean, Oceania, Europe and North America are predicted to remain relatively constant. Although future economic growth is projected to be less in developed versus developing countries, it is expected to continue moderately (Rosegrant *et al.*, 2009; Van Vuuren *et al.*, 2010). Concurrently, animal agriculture products will be marketed increasingly through global agribusiness supply-value chains (Rosegrant *et al.*, 2009; Chapter 12).

The total consumption of milk and meat is projected to be two- to approaching threefold greater in developing than developed countries

by 2050 and beyond, largely because of much greater population (Thornton *et al.*, 2006). However, annual per capita consumption of milk and meat in developed countries will still remain two- to threefold greater compared with developing countries. Global demand is projected to be strongest especially in South Asia and sub-Saharan Africa. Finally, rising global incomes will continue to increase purchasing power for animal products, especially in developing countries (Thornton, 2010).

This expanding population will increase demand for food, and doubtless put strain and stress on the resource base and almost undoubtedly increase risk of food insecurity. Certainly, with the current level of agricultural production, securing food for over 9 billion people (approximately two-thirds of which are projected to be urban dwellers) will be challenged significantly. Two critical aspects of agriculture science and animal production – to improve the efficiency of animal production (Chapters 2, 3 and VandeHaar and St-Pierre, 2006) and to increase the human-utilizable productive units per unit of resource use, while minimizing environmental and social costs during the production processes – will be critical.

Whether this growth in productivity and improvement in efficiency can be supported in truly sustainable animal agriculture production systems is a crucial and troublesome question. On the other hand, because it functions at a high level of intensity in more developed countries, animal agriculture has formidable challenges to meet the acceptable sustainability imperatives for people, planet and profit. That said, food production must remain profitable and food must be affordable for consumers. The most probable and simplest practical conclusion is that the food must be near where the people are located to be sustainable.

Production and supply

Production from animal agriculture increased dramatically in the last six decades or more with beef production more than doubling and poultry production increasing tenfold worldwide (FAOSTAT, 2010). These increases resulted from many more animals and perhaps more dramatically from increased productivity per animal and per farm. Harvested carcass weights increased by about 30% for both beef and poultry from the 1960s through the mid-2000s, and by about 20% for swine. Global average milk production per cow and egg production per hen each increased an estimated 30% over the same time interval (FAOSTAT, 2010).

These changes in animal agriculture have been associated with substantial expansions into areas and use of cultivatable land and pastures (Steinfeld *et al.*, 2006). Large conversions of forest land to crop and pasture lands occurred in the Amazon basin, Central and West Africa and South-east Asia, while at the same time there was some relinquishment of agriculture land in Eurasian boreal forest and parts of Asia, and the North and Latin Americas (GEO4, 2007). Major increases in cropland for soybean production (for oil and protein for humans and soybean meal for livestock feed) occurred in Latin America (e.g. in Brazil in the last 40 years). Additionally, the major increase in use of cereal grains (including maize) as animal feed occurred during this period in both developing but especially developed countries. The proportion of global use of cereals as animal feeds in developing countries nearly doubled (to 36% of the total) from the early 1980s to the late 1990s (Delgado, 2005).

Increases in animal production are associated with the types of (sub-) systems being used in different regions and countries. Confined animal systems in more industrialized countries account for much of the increased production of poultry and pigs, and similar systems have been installed in developing countries, for example, in Asia to meet increasing demand for animal meats. It is projected that at least three-quarters of total animal production growth by 2030 will be in confined animal systems (Bruinsma, 2003). However, much less growth of animal confinement systems will occur in Africa.

The expansion in animal production in developing countries is expected to occur primarily from escalating animal numbers, especially of ruminants (Thornton *et al.*, 2006). In intensive mixed plant–animal systems, food-feed crops (plants or their components that can provide either human food or livestock feed) have been essential feedstuff resources for ruminants. However, Rosegrant *et al.* (2009) predict that prices for food-feed crops are likely to increase at a rather more rapid rate than the prices of

animal products. Thus, the extensive utilization of food-feed crops as feedstuffs in animal production, and especially for ruminants, likely will be much less in the future. However, production of forages and concentrates is projected to vary widely among regions (Herrero *et al.*, 2009). Increased production of feed concentrates (e.g. maize and sorghum) may occur in sub-Saharan Africa; however, forage production for ruminants is not expected to expand in South Asia due to land-use competition from non-agriculture sectors (e.g. urbanization). Supply of alternative feeds (co- or by-product feedstuffs from other agriculture processing industries) for ruminants in more intensive mixed production systems may be challenged by scarcity of both land and water availability (especially for irrigation) (Herrero *et al.*, 2009). Also, consequences of climate change through time likely will shift locations and increase the temporal variability of forage and concentrate feed production (Strzepek and Boehlert, 2010).

Meeting major increases in demand for food worldwide will have significant impacts on animal production systems in the foreseeable future. In developed countries, increased productivity and the efficiency of production coupled with greater animal harvest weights will account for much of the growth, whereas increase in animal numbers is not expected to be significant, and may even shrink in some regions (Thornton *et al.*, 2006). However, compared with recent estimates, by 2050 the world's cattle population is projected to increase 1.7-fold, whereas sheep and goat numbers may increase by nearly 60% (Rosegrant *et al.*, 2009). Also, the grazing intensity on available grass- and rangelands is expected to increase, resulting in considerable intensification of animal agriculture in humid and sub-humid regions such as Latin America (Thornton *et al.*, 2006).

As the consequence of the factors noted above, the projected demand, supply and prices of animal products and food-feed crops are expected to increase in the future, contrary to past trends (Thornton *et al.*, 2006). The greater demand for animal products likely will raise prices of feed concentrates (e.g. maize grain and soybean meal) for livestock. Competition for land and water resources for production of biofuels from crops already is happening in some regions (Thornton, 2010), exacerbating availability of traditional feedstuffs and altering types and prices of feedstuffs available for animal agriculture. Overall, growing challenges for land and water use will necessitate significant improvements in efficiencies of resource use in animal agriculture to meet demands and avoid risk of food insecurities. Doubtless, changes in practices and management of animal production systems in the future will involve major trade-offs among people, planet and profit – food security, poverty, equity, environmental sustainability and economic development (Thornton *et al.*, 2006). Greater prices for animal products can benefit farmers producing extra commodities at least temporarily. Unfortunately, however, they also limit accessibility to food for a greater number of poor people, including poor livestock farmers especially in developing countries, who cannot produce net surpluses of food for local, regional or global markets. Thus, Rosegrant *et al.* (2009) predict that progress toward reduction of food insecurity and malnutrition will be slow at best.

Challenges and Competition for Resources

As emphasized time and again in this book, the 9 billion people projected to inhabit the earth by 2050 is the major challenge, especially when coupled with the fact that the vast majority of the additional population will inhabit urban areas in Asia and sub-Saharan Africa (United Nations Population Division, 2012). Also, supply, availability and use of resources are formidable challenges. Amongst the most worrisome, especially as related to the potential for a future sustainable animal agriculture, are the use of water and land, the competition between humans and animals for food versus feed, effectively building and managing nutrient and energy (e.g. organic residues) cycles and waste, and climate change, all interwoven within the influences and socially acceptable beliefs and standards of different societies and cultures (Orr, 2002).

Water use

Water is placed first in this section addressing challenges to emphasize that without sufficient

water in agriculture little else matters – including use of land, energy, improved or favourable biodiversity, or reduced contamination of water and air. Freshwater is an increasingly scarce resource in many regions of the world (Pimentel *et al.*, 2004; Rosegrant *et al.*, 2009). Among many other uses, it has an especially important role in animal agriculture and sustainability. However, it has not been studied and assessed as much in animal systems compared with plant systems. Water obviously is essential for all life. However, different organisms (e.g. animals versus plants) and different production systems dictate vastly different quantities (footprints) and efficiencies of water use (Rosegrant *et al.*, 2009; Hanasaki *et al.*, 2010; Hoekstra, 2010). Although water is no more precious to animal than to plant agriculture, some current animal production practices and systems especially in developed or rapidly developing countries will not be practical or even possible in sustainable animal agriculture systems in the next 50 years. It is widely known that production of animal products in some systems is very water-intensive (Chapagain and Hoekstra, 2003; Pimentel *et al.*, 2004; Raman, 2006; Herrero *et al.*, 2009). About two-thirds of all human water usage between 2000 and 2010 was to irrigate crops for food-feed production (Strzepek and Boehlert, 2010). Of that, about one-third of the total water footprint was associated with the production of animal products, mainly utilizing water to grow livestock feed (Mekonnen and Hoekstra, 2012). Also, as a consequence of global warming, changes in availability and distribution of water supplies are occurring (Strzepek and Boehlert, 2010). From modelling efforts, Strzepek and Boehlert (2010) predicted an 18% reduction in global water availability for agriculture by 2050. This will result in large part from the environmental flow requirements (freshwater needed to maintain natural riparian ecosystems), plus increased demands for municipal and industrial uses (two–fold increase in developing countries), and altered timing, distribution and supply of water due to climate change. The combined effects are projected severe water shortages in much of Asia, northern Africa, parts of Europe, eastern Australia and much of the western half of the USA.

Although the world's agriculture system will need to produce more food for an expanding and presumably wealthier population in the decades ahead (Steinfeld *et al.*, 2006; Janzen, 2011), the increasing demands for freshwater and potential impacts of climate change represent formidable threats to food and feed production for humans and animal agriculture. This will be further accentuated and stressed by practices in some animal production systems that are very water-intensive and likely will make these practices unsustainable in the not too distant future.

Recent research by Mekonnen and Hoekstra (2012) made a concerted effort to characterize and project the water footprints of various human-edible food products (of both plant and animal origin) on a nutritionally equivalent basis (protein, fat and energy), and also to consider the amounts and compositions of feedstuffs consumed per animal species in different farming system types (grazing, mixed and industrial) in four countries (China, India, the Netherlands and the USA). Water footprint was defined as the amount of water required to produce a unit of animal product from farm to fork, including water requirements to grow the different feeds that the animals consumed. The predictions also were compared with footprints for production of several plant crops. As a general conclusion in terms of use of freshwater, it was more efficient to obtain calories, protein and fat from production of crops than from animal products. The water footprint of any animal product was greater than the water footprint of any edible crop on an equivalent nutritional (protein or caloric) basis. For example, the most dramatic contrast was the water footprint per calorie of beef at about 20 times greater than for cereals and starchy roots. The water footprint per gram of edible protein from milk, eggs and chicken meat was about 1.5 times greater than for dry beans. However, the more specific nutritional quality of the protein (e.g. the profile and amounts of the dietary essential amino acids from different animal or crop food products) and fats (e.g. dietary essential fatty acids) was not considered in their analysis. Perhaps this would be a useful characterization and comparison, particularly applicable for parts of the world where deficiencies of high-quality dietary protein are a serious problem.

The overall greater water footprints of animal compared with plant products are due in significant part to the unfavourable conversion

efficiency of plant-derived feedstuffs to animal products. For example, among animal products evaluated the water footprints of meat production increased more than 3.5-fold; from chicken meat (4300 m^3 t^{-1}), to goat meat (5500 m^3 t^{-1}), to pig meat (6000 m^3 t^{-1}); and sheep meat (10,400 m^3 t^{-1}) to beef (15,400 m^3 t^{-1}). This wide range is explained partially by differences in feed conversion efficiencies. Pooled across the production system types (grazing, mixed crop–animal and industrial), eight times more feed dry matter was required per unit of beef meat produced compared with pig meat, and 11 times more than for chicken meat production. Besides the poorer feed conversion efficiency, obviously the composition of the diets (e.g. more fibrous feeds for ruminants, which are not utilizable by non-ruminants) differed appreciably among the animal species and system types. The 'overall global diet' was composed of a much greater proportion of concentrate feeds for broiler chickens (73%) compared with beef cattle (only 5%).

Additionally, from a freshwater perspective, animal products from grazing systems had smaller footprints than from industrial animal systems mainly because of less use of ground- and surface water. Also, within the four countries examined, less freshwater (called grey water) was required in the grazing system compared with the mixed or industrial systems to sufficiently assimilate (dilute) pollutants to meet water quality standards. Overall, model predictions clearly showed that it was a more efficient use of water to produce edible calories, protein and fat through crop production than through animal production (Mekonnen and Hoekstra, 2012).

Pooled across animal production system types in the four countries, total water footprint (the sum of precipitation, surface, ground and grey waters) per unit of product for meat production was greatest for grazing, then the mixed crop–animal system and least for the industrial systems. However, for dairy products the mixed system had the lowest (20% lower) water footprint compared with grazing or industrial dairy systems having quite similar, but larger footprints. The primary reason for greater water footprint from grazing, to mixed, to industrial systems for meat production is the poorer feed conversion efficiencies in systems where feeds contain much more structural, fibrous carbohydrate (e.g. cellulose, hemicellulose and lignin) and less non-structural carbohydrate (e.g. starch). In their models, three to four times more feed dry matter was required per unit of product principally due to difference in the utilization of the carbohydrate fraction of the diet and consequent poorer animal performance (Mekonnen and Hoekstra, 2012).

Within dairy products (milk, butter and cheese), the mixed crop–animal system resulted in the smallest water footprint compared with the grazing or industrial systems. This resulted from greater feed conversion efficiency of dairy animals consuming a mixed ration of forages and concentrates combined with less water requirement to grow the proper mixture of feeds (Mekonnen and Hoekstra, 2012).

Overall, more feed required for greater animal productivity requires more water to produce the feed, increasing the water footprint to produce the animal products. Of the total global water footprint to produce crops, about 20% is specifically for feed crops for livestock. In other terms, it is estimated that about 12% of the total global consumption of surface and groundwater for irrigation is utilized to produce feed for livestock, but not for food, fibres or other crop products for direct human use. Additionally, more water is required to produce concentrate feeds (high in non-structural carbohydrates) than roughage feeds (high in structural carbohydrates or fibre). In the case of industrial beef production (e.g. feedlots with high concentrate feeding), the greater footprint comes predominantly from the large amount of water (e.g. via irrigation) to grow the high-concentrate feeds for finishing cattle rations. Also, Mekonnen and Hoekstra (2012) noted that the industrial system added extra excretory nutrients and required more ground- and surface-waters for dilution to meet nutrient load standards compared with animal products from mixed-crop–animal, or grazing systems where the nutrients could be recycled directly to the soil. They speculated that as per capita global wealth and demand of meat consumption continues to rise in coming decades (Steinfeld *et al*., 2006), the continuing intensification of animal production will progressively exacerbate use of the world's dwindling freshwater supply (Mekonnen and Hoekstra, 2012).

Land use

Global animal agriculture occupies the largest land mass of any human activity (Steinfeld *et al.*, 2006; Steinfeld and Wassenaar, 2007). As much as 30% of the world's usable land is grazed, and about a third of the cultivated area is used for animal feed and forage production (Asner *et al.*, 2004; Ramankutty *et al.*, 2008). Little increase in available pastureland is anticipated in the future (Bruinsma, 2003; MA, 2005). However, some intensification and effective management of pasture-based production in sub-humid and humid areas is expected. Effective management of agriculture land in mixed plant–animal systems potentially can be quite sustainable and efficient if done appropriately in nutrient cycles (Shutt, 1913 as cited by Janzen, 2011). Improved utilization of land by ruminants in semi-arid and arid regions may be a potential option if managed properly (e.g. Janzen, 2011; Savory Institute, 2012). Issues with land use and animal agriculture can and do arise when excess nutrients from animal systems contaminate other lands, waters or ecosystems, or when the animals and land base are separated too far from one another for cost-effective recycling of nutrients among soil, plants and animals. In the future, especially more grazing systems likely will be positioned to provide ecosystems services and goods that societies demand. Perhaps animal agriculture in these situations also can be a benefit with tradable attributes (e.g. sheep or goats grazing public-urban easements for vegetation control). Especially in more developed countries, competition for land for biofuel production prompted by concerns over energy insecurity, climate change and alternative income sources likely will diminish land use for animal agriculture. However, it is noted that insufficient knowledge about the opportunities, options and interrelationships among food, feed and fuel in both developed and developing countries currently exist, especially with regard to coming second-generation bioenergy technology (Van Vuuren *et al.*, 2010). Additionally, destruction of Amazon tropical forests (to make grazing land for livestock) has raised major concern (Herrero *et al.*, 2009; Barona *et al.*, 2010) and desertification of natural rangelands (Savory Institute, 2012) are major ecological concerns resulting in loss of carbon storage capacity, habitat and biodiversity.

Food versus feed for humans and animals

Recent estimates are that between 30% and 40% of cereal grains (food-feed crops) produced worldwide are fed in animal agriculture (Garnett, 2009; Godfray *et al.*, 2010). This could increase to almost half of the total cereal production if current trends continue (UNEP, 2009). In the last half century or more, this rising grain production occurred mainly in developed countries with intensification of irrigation, relatively inexpensive (government-subsidized) fossil fuel prices to grow cultivated crops, and government farm subsidy programmes that stimulated crop production with attractive profits for farmers (NRC, 2010). Consequently, with greater grain supplies at relatively inexpensive prices, highly digestible/utilizable cereal concentrates and some supplemental protein sources were abundantly available for livestock. This allowed much greater animal growth and intensification of animal agriculture in some regions (Gerber *et al.*, 2010), and diminished or disconnected interconnections among soil, plants and animals especially crucial for effective nutrient cycling and healthy soils (Naylor *et al.*, 2005). Thus, more fertile cropland that could be used to produce human food produced feed for animals. Additionally, significant quantities of food energy and nutrients from highly digestible plant material were lost in the relatively poor conversion to animal products. Therefore, the number of people who might have been fed per unit of land presumably declined compared with the possible scenario in which humans consumed the cereals directly. The United Nations Environment Programme (UNEP) (2009) suggested that constraining global meat consumption in 2050 to 2000 rates would provide enough grain for 1.2 billion humans. However, of course, making the grains available for humans by reducing consumption of meat does not guarantee that the grain would become available as food for hungry people somewhere else in the world (Stokstad, 2010). Also, with reduced availability of land resources (noted previously) whether or not continued intensification and use of cultivated or cultivatable land and water resources (also noted previously) can or should be used for production of grain and high-quality protein feeds for animal agriculture, doubtless will be a critical question

in the future as demand for human food increases with rising population. Another consideration of this assessment, not yet elucidated, is whether the nutritional quality (e.g. dietary essential amino acid profile) of plant-derived versus animal products for humans is similar or not, especially when evaluated per land and freshwater use, or environmental or climate impacts.

Nutrient excretion and broken cycles

Intensification of agriculture (both plant and animal) in many developed regions has resulted in major increases in amounts and fluxes of N from application of synthetic fertilizers and manures. Anthropogenic fixation of N far exceeded all other disruptions of natural nutrient cycles more than 40 years ago (Delwiche, 1970). Nitrogen fertilization of cultured crops and consequent emergence of reactive-N now far exceed all natural inputs into landscapes (Gruber and Galloway, 2008). Only a very small portion of this added N actually appears in food products (plant or animal) with much flowing from farm systems to air and water as ammonia, nitrate, nitrous oxide, or nitric oxide, perturbing natural ecosystem cycles until ultimately becoming inert-N_2 again (Galloway *et al.*, 2008). Animal agriculture is a major source of reactive-N loss (Erisman and Sutton, 2008). Typically, as much as 50–80% of N ingested by animals is excreted (Chapters 2 and 3). Globally the amount of N excreted by animals is roughly the same quantity as that in all synthetic N fertilizers used (Bouwman *et al.*, 2009). Fifty per cent or more of ingested N is excreted from feedlot cattle (McGinn *et al.*, 2007; Erickson and Klopfenstein, 2010) and dairy cattle (Kohn *et al.*, 1997; Wilkerson *et al.*, 1997; VandeHaar, 1998) and subsequently emitted as ammonia into the atmosphere. Improvement in efficiency of utilization of dietary N and more effective management of excreted reactive-N represent major challenges for animal agriculture in the future. Nutritional and management approaches are being studied and some reduction of N loss from feedlot systems is possible (Erickson and Klopfenstein, 2010). Animals also can excrete large amounts of P (Bouwman *et al.*, 2009). When in excess and not properly managed in the farm, P can move with surface water and contaminate natural and man-made waterways. Also, most P used in both plant and animal agriculture is from high-density sources (natural mined deposits or industrial chemical synthesis), transported and redistributed in different landscapes. When not managed properly it not only causes environmental damage, but also is lost irreversibly for potential future use in plant and animal agriculture (Van Vuuren *et al.*, 2010).

Energy use

Long ago, intensified 'modern agriculture' was recognized as 'the use of land to convert petroleum into food' (Bartlett, 1978). In developed countries, use of relatively inexpensive (government-subsidized) supplemental fossil fuel energy has had a very significant influence on the large increase in productivity of animal agriculture in the last 50-plus years. For example, Hillel and Rosenzweig (2008) estimated that about 35 kJ of fossil energy were needed to produce 1 kJ of feedlot beef. Doubtless, worldwide the use of fossil energy to produce livestock feed and animal products will require greater and careful examination as future global energy supplies and demands, forms of energy, and markets and governments' policies evolve (Anonymous, 2010). It seems entirely plausible, as recent trends suggest, that less fossil fuel will be available for animal production by 2050 and different forms and cycles of energy utilization should be explored both in developed and developing countries.

Climate change

Animal agriculture is identified often as a significant contributor to global climate changes (Chapters 9, 10) and also as a set of managed biological processes that is likely to be substantially affected by global warming in the future. Recent estimates are that ruminant enteric methane (with about 23 times the global warming potential of carbon dioxide) and nitrous oxide (about 297 times) from manure N (from both non-ruminants and ruminants) account for about 9% of CO_2-equivalents from anthropogenic emissions (Gill *et al.*, 2010). However, when changes in other emissions resulting from altered land use associated with animal agriculture are included, the estimate of the net effects is about 18%

(Steinfeld *et al*., 2006; McMichael *et al*., 2007; Gill *et al*., 2010). Climate change is predicted to have significant influence on animal agriculture in the future, though the severity of impacts likely will differ among regions. Animal agriculture in different regions could or will be affected directly, but differently by such factors as water availability, extreme weather events, drought and flood, and heat stress; and indirectly by changes in feed and forage quality and quantity, host–pathogen interactions and disease occurrences, and increased fixed costs (e.g. cooling of housing for animals) and variable costs (e.g. variability in feed and energy supplies) (Thornton *et al*., 2009). Doubtless, reductions in animal productivity will occur in certain regions (e.g. the present day sub-tropics and tropics) experiencing harsh climatic conditions inducing even greater heat strain. In contrast, in more temperate regions, agriculture productivity is projected to increase somewhat with rises of 1–3°C in local ambient temperatures (IPCC, 2007). Overall, with the projected increases in global animal numbers in the future (IPCC, 2007), greenhouse gas emissions also will rise. To date, research successes to reduce enteric methane production by more than a few percentage points have not been demonstrated (Gill *et al*., 2010; Martin *et al*., 2010). Furthermore, the ruminant digestive tract evolved over millions of years and methanogensis is the sink for excess electrons from ruminal fermentation. It seems highly unlikely that dramatic reductions in enteric methane emissions will be possible in the next 20- or 50-year time horizon. Moreover, it is noteworthy that greenhouse gas emissions per unit of mass of human-edible food (not considering potential nutritional quality differences of the foods produced) are typically greater for animal products than for grain products; for example in the UK, beef (16 kg CO_2-e kg^{-1}) versus wheat grain (0.8 kg CO_2-e kg^{-1}) (Garnett, 2009). Thus, reducing beef consumption, at least in more developed countries, is one possible avenue for reducing greenhouse gas emissions (Garnett, 2009; Popp *et al*., 2010).

Social and cultural influences

Animal agriculture is woven intricately into the social and cultural fabric in many countries, especially developing countries, as it was in the past for now more developed countries. Doubtless, different countries are and will be influenced profoundly differently by animal agriculture; and animal agriculture will be influenced differently in different developing and developed countries. Obviously, animal agriculture contributes greatly to food security and human health. For poor and undernourished people, particularly children, consumption of even modest amounts of animal products has major benefits on physical growth and mental development (Neumann *et al*., 2003). Animal agriculture's influence on human livelihoods via production of food and non-food (wool, hides) products for income in either organized or informal markets is well known especially in poor developing countries. Animals also may provide draught power for more intensive crop cultivation and harvesting. They also may supply manure nutrients (e.g. N, P and K) for plants as part of a mixed system with nutrient cycles noted earlier. In some cultures animals are financial or barter currency to help farmers save and accumulate assets, or trade or sell for other needed resources. Animals may serve as a tool to help diversify sellable assets or to reduce risk. In some cultures, animals have significant value in defining social standing and for building relationships (Kitalyi *et al*., 2005). In communities where animals have a central role they can be associated with positions of leadership and allow owners access to political, natural and financial resources. Inevitably the social and cultural status of animals in communities will continue to evolve – some very useful, some perhaps harmful.

Somewhat in contrast in more developed countries, for example in European agriculture, there has been recent stronger emphasis and financial support for agriculture holdings to secure and expand their traditional role of providing ecosystems services and goods (e.g. natural areas for wild animal habitat, nature preserves or managed wetlands) and access to rural areas; these efforts are expected to strengthen in the future (Burton *et al*., 2005; Deuffic and Candau, 2006). These alternative opportunities may have high potential value for agriculture and cultural heritage as public goods and services in the future.

Also, in more developed countries where standard of living (wealth) and food sufficiency are greater, ethical concerns and causes are increasingly focused on roles, practices, management and functions of animal agriculture in society. Because of this, animal production and consumption of animal products already are being affected to varying degrees in some societies. Direct and indirect ethical considerations related to animal welfare, animal rights, veganism, vegetarianism, environmental veganism and organic farming are evident (Swanson *et al.*, 2011; Thompson *et al.*, 2011). However, overall there is a paucity of sound data and rigorous analyses to help inform the public discussion and policy-making processes (NRC, 2010). An example of this is the ongoing animal welfare research in the UK for more than 40 years (Brambell, 1965), with somewhat later initiation and development in North America (Chapter 4; Mench *et al.*, 2011), to gain a better understanding of animal behaviour and feelings, and how this body of knowledge should be incorporated into practices in animal agriculture production systems, whether overtly through public policy making or more indirectly via consumers' purchasing preferences and decisions (Waterman, 2008).

This is not to suggest that animal welfare is not a very important ethical consideration in some societies without strong economic development by modern standards. For example, the Massai culture of Kenya is an example often noted where animals are highly valued in the centre of existence and the primary source of livelihood (food, and as assets and insurance) (Kitalyi *et al.*, 2005). In any case, improving animals' welfare in agricultural production systems will not be an unnecessary expense and improved practices will serve as avenues to attract consumers, improve animal productivity in many cases and improve returns; although it is quite unclear how highly valued animal welfare considerations are to the majority of consumers, especially in developing compared with developed countries when considered against the cost of food. The complex interrelationships among social-ethical considerations (such as animal welfare) will require open dialogue with consumers as part of the formative process to determine practices and management influencing animal agriculture (Swanson *et al.*, 2011; Chapter 4). The considerations and acceptable standards regarding ethical issues will vary among countries, yet be influenced to significant degree by those participating in global markets.

Part of the increased social conscience related to animal agriculture is focused on study, development and adoption of new technologies. Doubtless, this is and will be part of the dialogue related to animal agriculture in the future. Considerable evidence indicates serious disconnect between science and public perception, spawning misconception and distrust. One of the key reasons for distrust is the paucity of credible, transparent and effectively communicated risk analyses associated with many highly technical issues and new opportunities. The methods, rigor and communication of good science will be crucial to ensure food security and improved well-being for 9 billion people by 2050; especially given the major climatic, social and technological challenges previously discussed on the horizon. Technology will be critical for major realignment and redirection of various food systems globally. However, incremental development and implementation of new technologies will fall short if not put in the context that new technology possibilities must be contextual with building more knowledge, networks and capacity in an integrated manner (Kiers *et al.*, 2008). Discussion for transformation of thinking, research and technological development to integrated holistic systems are addressed subsequently in this chapter.

Social concerns could seriously risk even careful application of such new science and technology to provide enormous benefits to people, planet and profit. Potential new developments will benefit greatly from open, transparent dialogue with stakeholders as part of the formative process to understand risks and expectations (Swanson *et al.*, 2011). If this is not to occur, science must take into account fully the environmental and health consequences that might arise before the new technology is implemented (Thornton, 2010). This will prove very difficult to accomplish.

Advancements in Animal Science and Technology

What advancements in the animal sciences are possible and needed to meet the future demand for animal products and sustainable agriculture?

Thornton (2010) identified and discussed three major areas, briefly addressed below.

Genetic improvement

The genetic potential of animals obviously is the first critical factor controlling the prospect of improved productivity and production efficiency. Tremendous improvements in this area were made in the last half-century using conventional animal breeding and genetic selection techniques and programmes (Leakey *et al.*, 2009; Chapter 5), almost exclusively in developed countries. Simultaneously, the nutrient composition of animal products also evolved via genetic selection to meet demand. Selection techniques included specific breed selection or substitution, within-breed selection for specific phenotypic traits (e.g. rate of gain, carcass quality traits, or milk yield and composition) and crossbreeding. Overall on-farm genetic progress typically ranged from 1% to 3% per year for single or multiple trait selection in both poultry and swine breeding programmes (Smith, 1984). Similar progress was achieved in dairy cattle in Europe, North America, New Zealand and Australia using sophisticated progeny testing programmes and artificial insemination (Simm, 1998; Simm *et al.*, 2004). Rate of genetic improvement in beef cattle and sheep, while substantial, was less with less aggressive use of progeny testing and artificial insemination. Whereas most of the genetic progress was made in developed countries, there is considerable potential to employ these same traditional genetic selection techniques in developing countries if the most important traits are identified and sufficiently heritable, and appropriate data recording and evaluation programmes can be developed. Some desirable and more important traits likely are quite different among developing countries with quite different climatic and environmental challenges than have been the focus in developed countries in more temperate regions. For example, moderate to high yielding dairy cattle breeds selected for temperate conditions are not sustainable in harsher environments where environmental (e.g. heat stress) and feed resource challenges exist (King *et al.*, 2006). Crossbreeding with local cattle adapted to the conditions offers potential in those regions.

The recent and future use of new emerging methods for genetic selection of animals with desirable traits is especially exciting. New molecular genetic tools likely will accelerate the focus on important traits and the rate of improvement of multiple traits beyond productivity, such as targeted product quality, longevity, improved animal health and welfare, and resistance to environmental stressors. Chapter 5 of this book as well as Leakey *et al.* (2009) address various techniques, such as DNA-based testing for marker genes indicating desirable traits, transgenic animal selection and cloning, and all are thought to hold significant potential for the future. The rate of genetic progress should accelerate appreciably as well. For example, genomic selection is projected at least to double the genetic gain for milk yield of dairy cows by effectively reducing the generation interval compared with traditional selection techniques (Hayes *et al.*, 2009). Genomic selection is likely to revolutionize selection of desirable traits in animals amongst countries and lead to specialized genotypes that specifically fit local or regional environmental and market conditions. This should help make animal agriculture more sustainable in these regions.

Preservation of genetic diversity also is crucial to the broader future sustainability of animal agriculture. This has been raised as a concern of the narrowly focused breeding programmes in some developed countries (Chapter 5; Nielsen *et al.*, 2006; Van Raden, 2007). If animal agriculture is to contribute to demands for food and farmers' livelihoods in the future in both developed and developing countries, preserving genetic diversity of animals will be crucial so that animals of different genotypes can be selected and used in various and changing environments (e.g. due to climate change or different feed resources). Policy and institutional frameworks to preserve genetic diversity and sustainable use of the wide range of current breeds are needed so that animals can be selected that can be sustainable under various environmental and economic situations (FAO, 2007).

Nutritional improvement

In the last century, the nutrient and energy requirements of farm animals in various

physiological states were established fairly well (e.g. NRC, 2001). In developed countries there are a number of computerized systems employed to evaluate and formulate rations for both non-ruminants and ruminants and to match animals' nutrient and energy requirements to locally available feedstuffs. Current incremental, fine-tuning research largely is focused on refining these systems and building more understanding of the more mechanistic and dynamic nature of feed intake, digestion, rumen microbial protein production, tissue nutrient metabolism and animal performance.

A significant body of work remains to be completed better to establish the relationships among animal performance, body and milk composition, feed needs and quality, the excretion of nutrients and energy from animals, and the consequent production and environmental costs upstream and downstream of the single farm (sub-) system and regional aggregate of farms. This more holistic work is and will help improve the whole-farm net efficiency for capture of nutrients and energy by animals and help facilitate understanding and establish realistic expectations for consumers of what is possible and probable, and provide guidance for legislative and regulatory agencies.

In spite of the increased knowledge and application of nutrition advances in developing countries, the vast majority of the world's animals, and particularly ruminants in pastoral and extensive mixed plant–animal systems in most developing countries, experience episodic or permanent nutrient and energy shortages (Bruinsma, 2003). Poor animal nutrition, even before genetic potential, is likely the primary limitation in small farms in developing countries. Considerable research has been done to improve forage quality (digestibility), availability and preservation, use of trees and crop residues and use of dietary nutrient supplementation for animals. Because various sorts of mixed plant–animal systems are very common in many parts of the world, better integration and management of nutrient and energy cycles most likely can improve animal productivity and enhance soil fertility (McIntire *et al.*, 1992). Nutrients and energy serve multiple purposes as food, feed and fertilizer. In developing countries there would seem to be opportunities to intensify practices and management of inputs and tools derived from more high-input systems in developed countries. Similar mixed plant–animal systems have been and still are predominant and successful in many farms in developed countries.

Other factors likely will weigh more heavily on improvement of animal nutrition in the future. At the forefront is the continuing goal to improve feed conversion efficiency (VandeHaar and St-Pierre, 2006; Chapter 2 and 3). The ever-continuous volatility of margins for farmers doubtless will be greatly affected by swings in feed and energy prices. Thus, improved feed conversion efficiency is one approach to help achieve profitability in animal production. Increased public awareness and concerns about the use of feed antibiotics in animal agriculture will become increasingly important also (Vallat *et al.*, 2005). The global trend was set in 2006 to reduce use with removal of all growth-promoting antibiotics from animal feeds in the European Union (HCPC, 2005). Some other developed countries likely will follow with similar actions in the future. Other exogenous chemical compounds (e.g. hormones influencing growth or reproduction) likely will be examined carefully and may have similar fates over time. In other cases, the precautionary principle simply may be employed until such time and resources can be invested to quantify relative risk and safety of any new or existing technology or performance-enhancing compounds (Recuerda, 2008). The vastly increasing globalization of markets for animal products (Chapter 12) will have major influence to heighten awareness and concerns about use of exogenous chemicals, food safety and food quality, and to affect the sustainability of animal agriculture in various regions.

Another key influencer affecting delivery of animal nutrition will be the need and (or) legal obligation to reduce air emissions (e.g. ammonia and greenhouse gases) originating from animal farms. Improved feeding practices (e.g. improved digestibility of forage fibre by earlier harvesting or the use of more highly digestible concentrate feeds) reduce enteric methane emissions. Reducing methane emissions per unit of edible animal product will be a key reference standard within and across animal species, animal agriculture system types and countries in the future. A significant effort is underway worldwide to discover potential dietary chemicals (including

antibiotics, probiotics and propionate precursors like fumarate or malate, or other nutrients) that would alter the ruminal ecosystem to reduce methane production (Smith *et al.*, 2007; Gill *et al.*, 2010). However, methanogensis is a fundamental disposal sink for excess electrons resulting from normal ruminal fermentation. Finding alternative electron acceptors, approaches or pathways to rid the ruminal ecosystem of excess electrons, yet maintaining near-normal ruminal fermentation, will prove a very formidable task.

Animal health

Overall, in the last several decades there has been a general reduction in the severity and prevalence of animal diseases, attributed mainly to more effective vaccines and pharmaceuticals and advancements in diagnostic strategies (Perry and Sones, 2009). Even so there were serious outbreaks of foot-and-mouth disease in the UK (Bio-Era, 2008), and of avian influenza H5N1. Each caused considerable global concern with the potential risk for shift in host from poultry to man, and emergence of a new human influenza, which did not occur. In developing countries over the last few decades, fortunately, there were few changes in the distribution, prevalence or severity of many epidemic or endemic diseases in animal agriculture (Perry and Sones, 2009).

In the future, occurrence and transmission of infectious diseases likely will be influenced by travel, migration and trade – all part of ongoing and accelerating globalization – that will increase risk of introduction of infectious agents into naïve animal populations. Confined animal management in large-scale systems and transport over greater distances can increase risk of disease transmission among animals (Otte *et al.*, 2007). Confounding in the longer time frame, disease trends could be modified significantly by changes in global climate. Geographical incidences of diseases may shift depending on the climate and suitable vectors. The nature and degree of impact of climate change likely will be different on future disease situations with livestock compared with known situations (Woolhouse, 2006). However, characteristics of future disease situations are not very predictable when influenced by: changes in overall climate in different regions, variability in climate, demographic changes in the relationships among animal and human populations, and deployment of effective technologies to survey and combat infectious diseases (Thornton *et al.*, 2006). Future changes are uncertain, at best.

Back to the Future

Holistic integrated mixed plant–animal systems

Animal agriculture in its early evolution in now developed countries was a crucial component of mixed plant–animal systems. Effective management of agricultural land in mixed plant–animal systems can be quite sustainable and efficient if done appropriately in well-coordinated nutrient cycles. In the past, having animals as key components of mixed plant–animal systems was advocated widely to maintain soil fertility (Shaw, 1911 and Shutt, 1913, as cited by Janzen, 2011). Shutt (1913) wrote, ‘How, then are soils to be maintained in a productive state and at the same time yield a profit for their working? First in the keeping of livestock; in the manure so obtained we have the opportunity of restoring to the soil eight-tenths of the plant food taken from it in crops they consume ... We do not keep sufficient livestock on our farms.’ In developing countries and most areas of the world where synthetic fertilizers are very expensive and (or) not available, recycling of nutrients through animals continues to be an indispensable practice, especially with use of N_2-fixing forage legumes (Wilkins, 2008). In the future, as high energy costs limit the production and use of synthetic fertilizers even for developed countries, having animal agriculture tightly linked with crop production in nutrient cycles can provide a natural way of using, recycling and re-using nutrients and organic residues (energy) effectively and efficiently. Also, beyond direct on-farm nutrient cycling from livestock feed to plant nutrients, animals (especially ruminants) have an additional advantage of being able to extract valuable nutrients and energy for by-product feeds

and food-processing wastes (Garnett, 2009; Bremer *et al.*, 2010). Then the animal waste can be recycled to soil; thus, creating an efficient loop where the same nutrients can be recycled over and over again, supplying human-edible food in each turn of the cycle.

However, in the latter part of the 20th century a notable (and well-publicized) portion of animal agriculture followed a model more akin to traditional industrial manufacturing, rather than one built predominately to ensure effective cyclic flow of nutrients and energy (Naylor *et al.*, 2005). For the most part, the industrial manufacturing process is not cyclic, but rather essentially straight-line, in which products are manufactured from raw materials into finished products very efficiently with the aim of minimal waste. In agriculture, the industrial model afforded significant improvements in labour utilization and land productivity during early phases of use (after which soil health was jeopardized and depleted), coupled with technological advances, increasing the economic value of agricultural products that could be realized. This evolution was heavily influence by changing markets and marketing systems (Chapter 12), and at the same time in countries such as the USA government-subsidized inputs (e.g. fossil fuel) and crop commodity programmes favourably affected both input costs as well as output revenue for some commodities (e.g. maize, soybean, wheat, cotton), aiding and even catalysing many animal farms to became more specialized, increasing economic efficiency per animal and per farm, and farm income (Hoshiba, 2002).

However, in contrast to the industrial model, animal agriculture by its very nature as a collection of biological processes is relatively inefficient at transforming raw materials (e.g. dietary nutrients) and energy into human-desired products (Hoshiba, 2002; VandeHaar and St-Pierre, 2006). Significant amounts of unutilized nutrients (e.g. minerals, organics and water) are generated during the transformation processes of feed to animal products. For example, the efficiency of dietary N capture in milk ranges from 20% to 30% with the remainder being excreted in manure (VandeHaar, 1998). Animal production in some operations is dependent upon large imports of feed grains and supplements and, in many cases, forages from local, regional farms, or even from crop-producing areas considerable distances from the animal farms. Under these circumstances, resulting manure nutrients especially if still in liquid form are not valued highly as a resource, but rather as a waste product that must be managed and ultimately disposed of (Chapters 7, 8 and 9). The once cyclic nature of nutrient utilization and management characteristic of mixed crop–animal farms to some extent gave way to a much less efficient version of an industrial manufacturing model with major waste disposal and management challenges (Chapters 7, 8, 9 and 10). The subsequent outcomes of industrial animal agriculture led to numerous instances of environmental problems without opportunity or sufficient ability to manage nutrient cycles or appropriate attentiveness to the general sustainability of the overall food production system (Hoshiba, 2002). This has resulted in considerable undesirable and unfavourable publicity especially for animal agriculture production systems.

There is a significant and sufficient body of evidence indicating that integrating (mixing) animals with plants provides much greater opportunities to maintain sustainable agro-ecosystems over time (e.g. Janzen, 2011). If properly managed, the requisite nutrient cycles can be sustainable; and especially important soil tilth can be continuously improved and maintained, which is paramount to any agriculture production system – especially, the sustainable mixed plant–animal system.

Figure 18.3 conceptualizes how plant and animal agriculture in both developing and developed countries might move towards the centre in a continuum of more sustainable mixed plant–animal systems for 2050 and beyond. The last section of this chapter addresses the necessity of challenging primary issues and paradigms, and capturing opportunities to catalyse transformation from the very extensive and from the excessively intensive farming systems to evolve towards a continuum of sustainable mixed plant–animal systems, of which animal agriculture will be an indispensable component.

Transformation to sustainable animal agriculture

Transformation of many sectors of animal agriculture in developed and developing countries

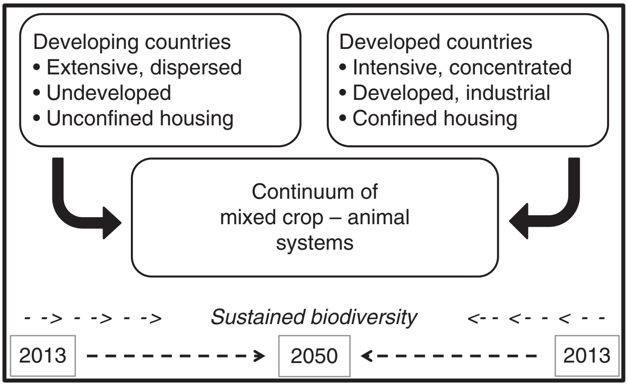

Fig. 18.3. Convergence of the continuum of future sustainable mixed crop–animal farming systems with appropriate development and implementation dependent on local resources and objectives in developing and developed countries.

to more sustainable mixed plant–animal agroecosystems during the next 50 years will require enormous resolve, commitment and work. The report by the NRC (2010) expert committee, *Toward Sustainable Agricultural Systems in the 21st Century*, provides an analysis and conceptual framework for a determinative set of objectives and processes for re-orientation of agriculture in the USA and other developed countries, and for developing countries such as those in sub-Saharan Africa. Transformational development will require major investments of both social and economic/financial resources by both the public and private sectors, throughout the value chains of production agriculture and the food industry.

The challenges and opportunities for animal agriculture particularly are emphasized in the present chapter. The NRC (2010) report emphasizes that agriculture and societies can benefit greatly from major shifts in thinking and practices towards extensive application of the principles of sustainable agriculture. As noted in the introduction of this chapter the committee named four key imperatives that must be adopted and actively pursued for an acceptable degree of agriculture sustainability to be approached and achieved in the future. In short, these four imperatives are to provide a secure, safe supply of food for people, to protect and enhance the quality of the environment of the planet, to foster and support the economic viability, vitality and profit of agriculture, and to enrich the quality of lives of people. The adoption of potentially sustainable farming practices and (sub-) systems must be assessed by how well they satisfy these imperatives and other relevant societal objectives. To be sustainable a practice or (sub-) system must be sufficiently productive, robust (e.g. be able to continue to meet sustainability goals under various challenges), use resources efficiently and effectively balance the four imperatives in socially acceptable ways. Doubtless, progress toward sustainability of animal agriculture as part of mixed plant–animal systems will require substantial shift in the visions among most participants directly and indirectly involved in agriculture and from society. Significant intentional redirection and redeployment of investments in research, education and outreach will be necessary. Also, as part of the transformation process transparent communication and participation among farmers and all in agriculture, the public and private sectors in evaluation and demonstration of a variety of potentially sustainable practices and farming systems will be necessary and crucial.

The committee report advanced and discussed two main approaches to enhance prospects for continuous improvement towards sustainable performance of agriculture, including animal agriculture: (i) incremental work to find successive improvements of varying magnitudes, to increase quantity or enlarge value; and (ii) a transformative approach for a major paradigm shift (NRC, 2010). The incremental approach (currently the main approach employed by most federally and state-funded programmes and projects, for example in the USA) focuses

on acquiring new knowledge, development and implementation of very specific concepts and practices associated with production strategies and (or) environmental concerns almost exclusively for current mainstream, conventional farming systems. Examples from animal agriculture include how to utilize dietary nutrients and energy more efficiently, or to reduce the excretion of methane and N, or how to improve reproductive management of a herd. Whereas these are important issues in the present systems and could benefit from improvement, they frankly are not currently the most urgent challenges for long-term sustainable animal agriculture. Incremental improvements or mitigation of negative outcomes of animal-biological processes are useful, but do not necessarily help us rethink or redesign whole system(s) towards improving sustainability performance across many farms, regardless of size or farming methods. These individual discoveries and practices are rarely considered and studied in the more comprehensive integrated 'systems context' (Fig. 18.1). Only later, after implementation, are unintentional consequences oftentimes discovered. Although the incremental approach offers discovery and demonstration of useful specific practices, it is not adequate or appropriate to address multiple, complex sustainability concerns.

The report stressed that more research must address the multiple dimensions of sustainability and understand agro-ecosystem properties and interrelationships if systemic changes in farming systems are to be pursued (NRC, 2010). Thus, the incremental approach, perhaps helpful to discover single or a few elements or strategies to improve sustainability, must be supplemented and greatly reinforced by what the committee termed this 'transformative approach'. This approach would enhance greatly integrative exploration of sustainability by multidisciplinary efforts exploring well beyond agro-ecological dimensions. The transformative approach considers ways to change the whole system, rather than simpler singular, one-at-a-time incremental developments. Transformative examples in animal agriculture might include alternative mixed plant–animal agriculture systems (e.g. grass-fed-finished beef value chains, combination organic milk and dairy-beef production, or poultry-catfish farms). The transformative approach must employ rigorous systems science approaches (NRC, 2010). This will be essential to build the understanding of agriculture as a part of the larger complex socio-ecological system(s), for which research must identify and understand the importance of the linkages among farming components, the resource base and societies (Fig. 18.1), and how their interconnectedness and interrelationships with the environment make systems strong, adaptable and harmless or even beneficial to the environment and society through time. This is not a new conceptual approach (Douglass, 1984; Lowrance *et al.*, 1986).

Thus, the appropriate and relevant incremental efforts simply would become a sub-set of efforts embedded within the holistic systems-based transformative framework. This would elevate the crucial nature of systems thinking to the highest order for long-term development of sustainable agriculture, including animal agriculture. To pursue this change, the committee proposed major re-orientation of publicly funded agriculture research and education to better embrace and accelerate discovery and to direct development long-term sustainability efforts for agriculture (Reganold *et al.*, 2011).

However, more importantly, the committee also argued that currently, at least in the USA, the major impediments to rapid adoption of sustainable farming systems and innovative practices are lack of forward-thinking policy adoption and market structure challenges; much more than a need for more and new research and innovation. Progress forward to more sustainable systems is to a significant degree hindered by current market structures, misplaced or misinformed policy incentives, and uneven development and availability of scientific information to inform farmers' decisions about more sustainable systems (Fig. 18.4; adapted from Reganold *et al.*, 2011).

Overall, the committee identified three key drivers to shift the paradigm towards more sustainable agriculture thinking and doing, presumably including animal agriculture: (i) changes in markets; (ii) changes in federal, state and local policies; and (iii) changes in government funding priorities. The committee noted that current trends in marketing of agricultural products are largely driven by

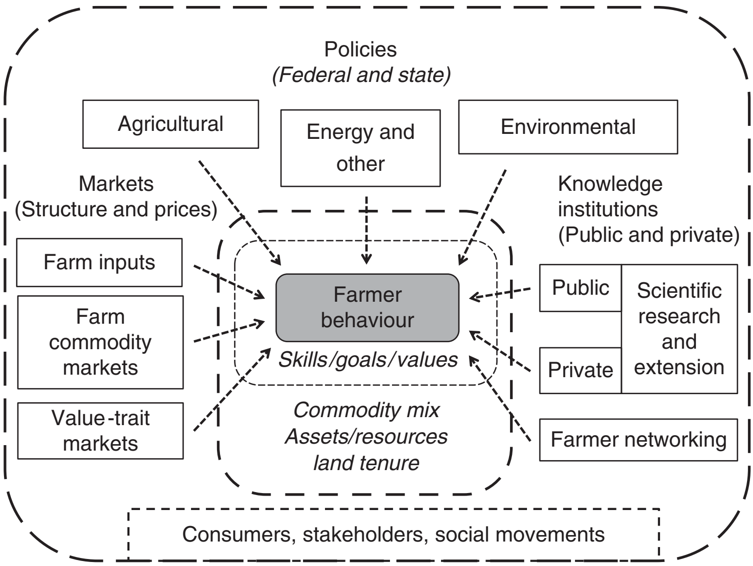

Fig. 18.4. Drivers and constraints that affect farmers' decisions. Farmers make choices based on market structures, policy incentives and knowledge institutions – all affected by consumers, stakeholders and social movements. Adapted from NRC (2010).

disproportionate incentives along supply chains to produce large quantities of relatively inexpensive commodities very efficiently.

Markets

Chapter 12 of this book provides extensive description and details of current agricultural marketing trends and global markets. The NRC (2010) committee recognized that most farmers especially in developed countries sell primarily commodities to a highly consolidated global market largely owned and managed by a relatively few entities in the global agri-food system that rewards large volume, low-cost food, feed and fibre, most often dictated and constrained by contract agreements from food processors and retailers. At the same time in developed countries, current-day consumers' buying and consuming habits have largely bolstered mainstream conventional farming (sub-) systems and food markets, contributing to obesity and health issues (NRC, 2010; Kratz *et al.*, 2012). Doubtless, a large part of the transformative process to establish sustainable agriculture systems must include education of consumers to change dietary habits and food preferences (USDA and USDHHS, 2010).

Consumer insistence for direct accountability of farmers for the environmental and social consequences of their farming practices and systems also has grown considerably in recent decades; for example, acceptable animal welfare and resource (water, land and air) conservation. Therefore, various 'sustainability branded' foods are appearing in grocery stores and restaurants as 'certified'. Markets, local, regional and even global, are responsive and becoming more responsive to these value-added products as driven by evolution to more sustainable systems, at least partially meeting the sustainability imperatives set forth by the committee (NRC, 2010). These include grass-finished beef, organically produced milk and recognition of 'locally branded' products. While this movement has been late to arrive in the USA, but more advanced in the Europe, market forces that influence purchasing decisions likely will be accelerated via public-policy incentives that bolster demand for products from sustainable agricultural farms and (sub-) systems.

Policy

The committee noted that current, poorly formulated public policy (e.g. energy and farm

legislation) pertaining to agriculture is to a large degree responsible for the current agriculture production system in the USA (NRC, 2010). Many policies (international, federal, state and local) involving agriculture practices, credit, energy, risk management and environment dominate in farmers' management decisions (Figs 18.1 and 18.4). These policy drivers directly and indirectly influence single farmer decisions and aggregates of local or regional farming systems, greatly influencing the degree to which they operate sustainably. For example, in the USA, the best-funded provisions of the Farm Bill (revisited, revamped and renewed every 4–6 years) traditionally include: food-purchasing assistance to low-income families; significant commodity subsidies paid to farmers for production of maize, cotton, rice, soybean and wheat; crop insurance and disaster relief; and conservation programmes (Monke and Johnson, 2010). About one-third of US farmers receive commodity and conservation payments via the Farm Bill, which has major influence on what, where and how food and feed are produced and made available as human-consumables.

The committee also noted that most elements of the US Farm Bill traditionally are not intended to foster agriculture sustainability. Commodity subsidies are often criticized for distorting market incentives and making the food system overly dependent on a few crops (e.g. maize, wheat, soybeans), which are used currently in significant amounts by animal agriculture to grow and finish both ruminant and non-ruminants and for highly processed foods; both unfortunately, if not managed properly, can have damaging effects of the environment and human health (Dobbs and Petty, 2004; Cox, 2011). All of which is largely undergirded by relatively inexpensive government-subsidized fossil fuels in the USA compared with most other even-more developed countries.

The NRC (2010) committee argued that while significant remodelling of the US Farm Bill to foster sustainable agriculture will be an arduous task always with political friction and budgetary constraints, much of the knowledge and information needed to take the transformative path already is available (Batie, 2009). Proposed remodelling includes reduced spending on subsidies that mask reality and risks associated with conventional farming that facilitate problems of unbalanced markets, and cause issues with environmental and social acceptability – unsustainable agricultural performance. Funds should be re-deployed to encourage farming systems and markets that encourage and foster the four sustainability imperatives, and are more able to respond to resource constraints and variability in global markets.

In the broader policy view, the NRC (2001) committee encouraged much greater integration of the concepts, practices and systems of agriculture sustainability into policy discussions about many other looming issues such as biofuel production, energy policy, climate change, trade agreements, immigration reform and environmental regulation. Otherwise, it maintained that there would be very little possibility to achieve any major policy change or changes in farming practices or systems to achieve long-term agriculture sustainability.

Research and technology transformation

The committee also stressed publicly funded research and technology development should be re-directed and transformed in such a way as to directly and inclusively make agricultural sustainability the primary platform for identifying research needs and deployment of competitive as well as non-competitive local, state and federal funds. This would bring the transformational approach described above to the forefront and facilitate more integrated multidisciplinary projects and programmes. Currently, the majority of public and private agriculture research in the animal sciences is narrowly focused (the incremental approach) for enhancements in productivity and efficiency of production, especially on development or refinement of technologies that fit into existing production systems or aim to mitigate problems caused by these systems (e.g. water and air pollution). Also, often the more incremental discoveries and development lead to proprietary private technologies or benefits (Huffman and Everson, 2006; NRC, 2010). This may represent a misuse of

public funds and certainly may not be conducive to enhancing the transformational framework for developing and implementing widespread sustainable agricultural practices and systems. In the USA today, relatively little of public funding for agriculture, despite efforts of the USDA National Institute for Food and Agriculture, goes towards funding this proposed transformative approach, most goes to funding incremental research (NRC, 2010). Major reallocation of public funds is critical to support transdisciplinary systems research to examine the interconnected challenges and opportunities at the farm (agro-ecosystem) and landscape levels, as well as the complex socio-ecological system framework (Fig. 18.1).

Currently, as opposed to the vast portfolio of individual knowledge pieces available to farmers for decision making from incremental research, the movement toward transformative agriculture systems depends on a much smaller, developing knowledge base tested and communicated by farmers and non-profit organizations independent of traditional agriculture research institutions. Current agriculture research and farmers would both benefit from collaborative projects to build a publicly accessible database of innovations for sustainable agriculture, including animal agriculture. Additionally, re-deployment of some federal farm subsidies to fund, track and compare conventional systems with innovations from the transformative sustainable agriculture approach should be made at the landscape, watershed and airshed levels.

The NRC (2010) committee concluded its urge for the transformative approach to strive for sustainable agriculture by noting the tough choices that must be made among competing options and goals for a sustainable food system. It strongly urged that in addition to changing markets and policy, that substantive public dialogue is required to shape and direct the transformative path to sustainable animal agriculture systems. Successful implementation will necessitate the public, individuals and organizations spanning political and institutional boundaries to integrate complex components for transformation to agricultural sustainability from research, to field-testing and demonstration, to full implementation for farms, markets and consumers.

Thinking of mixed-crop–animal farming systems as the transformative target, the goals for agriculture sustainability are not exclusively applicable to either more developed or less developed countries or cultures. It is critical that animal agriculture sustainability be thought of as part of the transformative process in the context of people, planet and profit, rather than simply as a collection of incremental technical problems to be managed and solved. Systems thinking is required. Experiences and outcomes (non-sustainable as well as sustainable) over the last 100 years of development of agriculture systems in developed countries can serve to inform the evolution to sustainable systems in developing countries, and farmers from more developed countries can learn from long-held sustainable agriculture practices of less-developed countries, especially faced with less resources such as water and fossil fuel (Hillel and Rosenzweig, 2008). These will be paramount as evolution towards more integrated and sustainable mixed-crop–animal systems occur toward 2050 and beyond.

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